marine research specialists

3140 Telegraph Road, Suite A · Ventura, CA 93003 · (805) 644-1180

Mr. Bruce Keogh
Wastewater Division Manager
City of Morro Bay
955 Shasta Avenue
Morro Bay, CA 93442

3 March 2006

Reference: Responses to Technical Comments from NRDC[1,2,3,4] on the NPDES Permit[5] to be Reissued to the MBCSD[6]

Dear Mr. Keogh:

Pursuant to your request[7], we have evaluated the technical merit of the referenced comments. This letter focuses on the scientific and technical aspects of comments related to potential environmental impacts from the MBCSD discharge, rather than those dealing with the legal and engineering aspects of the discharge permit. Based on our detailed review, we find no substantive scientific merit in the assertions made by NRDC,[1] CEA,[2] Heal the Bay,[3] and Sierra Club.[4] Their comments consist largely of unsupported speculation and do not countermand the conclusions contained in the MBCSD monitoring reports.[8] Specifically, two decades of intensive and comprehensive monitoring has not found any deleterious environmental impacts associated with the discharge. Instead, the monitoring has demonstrated that the low-volume discharge of high-quality effluent rapidly disperses within an extremely limited area of the well-mixed open-ocean environment immediately surrounding the diffuser structure. There is no rational reason to believe that small incremental increases in flow over the next decade will cause a perceptible degradation in the marine environment.

QUALIFICATIONS

Marine Research Specialists (MRS) has been conducting the NPDES discharge monitoring program for the MBCSD for twelve years. Our qualifications depart from those of NRDC and their consultants. For example, CEA cites mostly engineering experience, Gold documents environmental advocacy, and NRDC staff are

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1 NRDC (Natural Resources Defense Council). 2006. *Time is of the Essence: The Legal and Technical Reasons Why EPA and the Regional Board Must Deny the 301(h) Waiver and Require Upgrade of the Morro Bay-Cayucos Sewage Plant "As Fast As Possible"*

2 CEA (Carpenter Environmental Associates, Inc). 2006. *Re: Morro Bay 301(h) Waiver*

3 Gold, M. (Heal the Bay). 2006. *Re: Re-issuance of the 301(h) Waiver for the Morro Bay-Cayucos Sewage Plant*

4 Sierra Club. 2006. *RE: Reissuance of a National Pollutant Discharge Elimination System (NPDES) Permit, Modified Under Section 301(h) of the Clean Water Act.* Via electronic mail to Mr. Matt Thompson, RWQCB, 2 February.

5 US Environmental Protection Agency Region Region 9 (USEPA) and the California Regional Water Quality Control Board, Central Coast Region (RWQCB). 2005. *Joint Notice of Proposed Actions on Reissuance of Waste Discharge Requirements* [NPDES Permit] *to Discharge to the Pacific Ocean for the City of Morro Bay and Cayucos Sanitary District San Luis Obispo County*. Public Notice No. RB3-2006-0019, NPDES No. CA0047881. 19 December.

6 The wastewater treatment plant jointly owned by the City of Morro Bay and the Cayucos Sanitary District (MBCSD)

7 Keogh, B. (personal communication). 2005. Telephone conversation between Mr. Bruce Keogh, MBCSD Wastewater Division Manager, and Dr. Douglas Coats, Senior Oceanographer, Marine Research Specialists (MRS) on 8 February 2006.

8 The most recent monitoring results are contained in the 2005 Annual Report, which is included herewith as an attachment.

largely focused on environmental litigation. None has specific experience in the evaluation of marine impacts, and this lack of experience is reflected in their inapposite comments.

MRS expertise lies in the quantitative assessment of marine impacts. Two MRS employees, Dr. Coats and Ms. Luke, are primarily responsible for the MBCSD monitoring program. For the past thirty years, Dr. Coats has specialized in the assessment of marine impacts based on analysis of multi-disciplinary data. He received his Ph.D. from Scripps Institution of Oceanography where his academic training focused on collection and interpretation of oceanographic data from virtually every aspect of marine science. Since that time, his quantitative assessments have resolved a variety of complex marine environmental issues by interrelating biological, chemical, and physical measurements. Many of his definitive assessments have been published in the peer-reviewed literature. Currently, he is the Program Manager for NOAA's assessment of long-term recovery of the intertidal habitat following the Exxon Valdez Oil Spill. More locally, he conducted a ten-month dye-tracking program within Estero Bay to monitor directional drilling associated with the installation of fiber-optic cables. The extensive drill-mud monitoring program was conducted on behalf of San Luis Obispo County, the California Coastal Commission, and the RWQCB. His experience specific to ocean wastewater discharges includes evaluations of NPDES monitoring data for the Los Angeles Clean Water Master Plan, the San Diego Metro Clean Water Project, Hyperion, East Bay Dischargers Authority, and the Massachusetts Water Resources Authority. He is the Program Manager for the MBCSD monitoring program.

Ms. Luke holds a BS degree in biological sciences from the University of California at Davis. Previously, she has worked as a freshwater ecologist for the Clear Lake Environmental Research Center, and spent a field season at the UC Davis Castle Lake Limnological Research Laboratory. She has also worked as a coastal program analyst for the California Coastal Commission evaluating development proposals for consistency with the California Coastal Act and other applicable local, state, and federal laws. She has spent the last several years as the co-program manager of the MBCSD monitoring program and is currently a senior marine biologist at MRS.

SUMMARY OF CONTENTS

RESPONSES TO SPECIFIC NRDC COMMENTS

1. MBCSD monitoring data, analyses, and reports are timely, comprehensive, and pertinent to the NPDES discharge permit. NRDC inappropriately characterizes the vast record of monitoring data as "*inadequate*," and "*stale*," and claims, therefore, that the discharge application is "*incomplete*."[9]

a) Contrary to NRDC's bizarre characterization,[10] data does not go "*stale*" as though it were day-old bread. In fact, the opposite is the case. Impact assessments, in particular, benefit from a long record of high quality observations because they provide crucial additional degrees-of-freedom that substantially improve the statistical power to detect marine impacts.[11] Spatiotemporal analyses of long-term trends are especially capable of definitively discerning subtle marine impacts when other techniques fall short. This type of analysis was applied to two decades of marine monitoring data in the most recent MBCSD Annual Monitoring Reports.[12] The findings of "*no deleterious impacts*" from the discharge were largely unchanged from the comprehensive analysis performed in the Technical Support Document (TSD), which was submitted with the NPDES discharge application.[13]

b) It is also greatly misleading for NRDC to imply that there has been no review of the MBCSD discharge since the permit application in 2003.[10] In fact, the MBCSD continues to submit monthly, quarterly, semiannual, and annual monitoring reports, which are matter of public record and are scrutinized by both EPA and RWQCB staff. NRDC also requested, obtained, and acknowledged the receipt of the 2004 Annual Monitoring Report, however, any mention of this report, or the results of the comprehensive analyses contained therein, is conspicuously absent from their comments.

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9 NRDC, at 2 and Part 3A

10 NRDC, at 4, alleges that the MBCSD did not submit a "*…complete application with current data and information…*," and that the "*…EPA's and the Regional Board's analyses, findings, and determinations are based on incomplete and stale information.*"

11 Coats, D.A., E. Imamura, A.K. Fukuyama, J.R. Skalski, S. Kimura and J. Steinbeck. 2003. Sampling Effort in Assessments of Oil-Spill Impacts to Intertidal Organisms. NOAA Technical Memorandum NOS OR&R 12, August 2003.

12 Marine Research Specialists (MRS). 2005. Offshore Monitoring and Reporting Program, 2004 Annual Report. Submitted February 2005 to the MBCSD; and MRS. 2006. Offshore Monitoring and Reporting Program, 2005 Annual Report. Submitted February 2006 to the MBCSD. Hereinafter "*2004 and 2005 Annual Reports*."

13 MRS. 2003. Supplement to the 2003 Renewal Application for Ocean Discharge under NPDES Permit No. CA0047881. Prepared for MBCSD. July. Hereinafter "*Supplement*."

Consequently, there is no basis for the NRDC statement that there is an "*…incomplete record that fails to include any reported violations of effluent limitations for nearly three years.*" All monitoring data and reports have been submitted on a timely basis as required by the monitoring and reporting component of the NPDES discharge permit.

c) Contrary to NRDC statements,[10] the MBCSD application for the NPDES discharge permit was both timely and complete. As required, the application was submitted six months prior to the expiration of the existing permit. It included detailed comprehensive analyses of more than a decade of monitoring data and encompassed quantitative observations that were collected up to the time of the submission of the application. Careful review of this vast amount of data requires time, and the existing permit was administratively extended to allow for a comprehensive assessment by the applicable regulatory agencies.

d) Finally, NRDC also asserts that the record is incomplete because "*…various consultation requirements…*" have not been met.[14] NRDC subsequently cites only one consultation with the US Fish and Wildlife Service (USFWS) as being out of date. In fact, an update to previous consultations with the US Fish and Wildlife Service (USFWS) was requested by MBCSD at the time the discharge application was submitted in 2003. Because of prior commitments, USFWS personnel could not provide an updated consultation at the time of the application. However, according to the USFWS,[15] formal interagency consultations with USFWS are not required under Section 7(a)(2) for all actions. Instead, each Federal agency is responsible for determining whether its action is likely to jeopardize the continued existence of any endangered species or threatened species or result in the destruction or adverse modification of habitat.

2. Wastewater constituents do not enter the Morro Bay Estuary in any ecologically meaningful amount. NRDC repeatedly states that effluent enters the Morro Bay Estuary, erroneously implying that significant concentrations of effluent regularly enter the Estuary.[16] They use this misleading statement to imply that, as a result, "*...the discharge…is highly correlated with the occurrence of…T. gondii*," and "*…the discharge…enters into estuaries where a balanced indigenous population does not exist*."

a) Contrary to their assertions, it is not "*well accepted*" that effluent "*…regularly enters and mixes with fresh water in the Morro Bay estuary,*" in any ecologically meaningful amount. It is not "*…undisputed that the…Plant discharges an average of 1.4 million gallons of freshwater wastewater into Morro Bay every day….*"[17] The MBCSD discharge is into Estero Bay, at a location well removed from the Morro Bay Estuary. NRDC base their assertions on a single dye-tracking study conducted on one day in 1985.[18] However, this dated and limited study provided no credible evidence that perceptible concentrations of effluent constituents actually entered the Estuary. The only meaningful discharge-related dye measurements were collected within 500 yards of the diffuser where the effluent had yet to achieve a dilution more than about 5000:1. Subsequent reevaluation of this study's data demonstrates that all of other the dye measurements consisted of background fluorescence associated with patchy, naturally occurring phytoplankton.[19] Effluent that was more dilute than 5000:1 contained dye

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14 NRDC, at 2, 21, 53, and 54.

15 Kofron, C. 2006. Personal communication between C. Kofron (USFWS) and B. Keogh (MBCSD) on 1 March 2006.

16 NRDC, at 2, 10, 13 38-39

17 NRDC, at 36.

18 Northeast Technical Services Unit (NTSU). 1986. Ocean Outfall Study, Morro Bay, California, October, 1985. Submitted to Region IX Shellfish Specialist, Department of Health and Human Services, Public Health Service, Food and Drug Administration, Center for Food Safety and Applied Nutrition Shellfish Sanitation Branch. August 1986.

19 Extensive dye monitoring within southern Estero Bay over a 10-month period in 2001 found that 0.5 ppb is the practical limit for reliably detecting dye concentrations against the background of naturally occurring

concentrations below approximately 0.5 ppb, which is the effective dye-detection limit in the open ocean. Nevertheless, the study investigators claim to have detected dyed effluent within the Estuary, after being diluted 50,000 times, even though the associated dye concentrations are virtually impossible to detect. Nevertheless, even if you accept their claim, one part effluent in 50,000 parts estuary water is so dilute that it cannot conceivably be considered ecologically meaningful.

b) Regardless, the ultimate far-field disposition of the effluent plume is irrelevant since the bacteriological quality at the end of the treatment process, and prior to discharge into the ocean, is almost always substantially lower than shellfish harvesting and water contact standards.[20] In addition, an accurate evaluation of potential impacts from wastewater particulates should be conducted after they have been diluted at least 100-fold. This would make any potential bacterial pathogens within the effluent virtually non-existent by two orders of magnitude.

c) Because of the rapid initial dilution that occurs shortly after discharge, it is highly unlikely that MBCSD effluent enters the Morro Bay Estuary or impinges on the adjacent shoreline in any meaningful concentrations. Even under unusually strong prevailing flow conditions, effluent is transported away from the outfall in an extremely diluted state. Quarterly water-quality monitoring surveys repeatedly demonstrate that the footprint of measurable amounts of wastewater is highly localized around the outfall. The highly limited spatial extent of the MBCSD discharge plume has been delineated with standard measurement techniques that have been applied during 28 quarterly water-quality surveys conducted over the past seven years of the current permit cycle. High-resolution water-quality measurements repeatedly demonstrate that the plume is diluted by several hundred-fold within 50 ft of the diffuser structure.[21] These dilution factors are consistent with those determined by conservative plume modeling[22] as well as the measurements near the outfall in the dye-tracking study cited by NRDC.[18] Beyond that distance, even the most sensitive instrumentation is incapable of detecting wastewater constituents. Consequently, it is highly implausible that perceptible concentrations of wastewater constituents would traverse a distance of more than 9,000 ft (1.7 miles) and enter the Estuary, even if strong southerly currents happened to coincide with a flood tide.

d) At the time of the 1985 dye-tracking study, the influence of matrix interference from planktonic chlorophyll was unknown. In the absence of control samples and without a standard statistical analysis of variance, background fluctuations in fluorescence were misinterpreted as dye concentrations. Additionally, none of the elevated dye concentrations, erroneous or otherwise, were continuously tracked directly from the discharge into the Estuary. Instead, the offshore dye-tracking tracking effort was concluded prematurely. On the following day, a fluorometer was deployed within the Estuary near the mouth of the harbor. Subsequently, extremely low levels of chlorophyll fluorescence were erroneously ascribed to the presence of effluent that had been diluted a phenomenal 50,000-times. Even if this fluorescence did represent the presence of wastewater constituents, at the measured concentrations effluent would be so extraordinarily dilute that it could not be of any conceivable environmental influence.

e) The 1985 plume-tracking study was originally commissioned to determine whether effluent discharge was causing coliform contamination in shellfish growing areas of the Morro Bay Estuary. At the time of

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chlorophyll fluorescence within the ambient ocean waters of the Bay. See Coats, D.A. 2003. Monitoring of Drilling Fluid Discharges to the Marine Environment of Estero Bay California during Construction of Directional Bores for the MCI/WorldCom Fiber-Optic Cable Installation Project. Final Report of Offshore Monitoring Results to the County of San Luis Obispo. MRS Technical Report MRS-252.

20 Prior to discharge, the effluent achieves receiving-water bacterial standards more than 90% of the time according to the historical record of measurements during this permit cycle.

21 Supplement, at III-2, and Table 3.8 in 2004 and 2005 Annual Reports.

22 Lindstrom, K. 1998. Report on Evaluation of February 1998 Sampling Results: AERA Energy LLC (Aera) Produced Water Discharge Proposal. K.P. Lindstrom, Inc. P.O. Box 51008, Pacific Grove Ca. In association with Carollo Engineers, 4200 Truxtun Ave, Suite 200, Bakersfield, CA 93309.

the survey, the MBCSD discharge was not disinfected prior to discharge. However, the bacterial measurements that were collected offshore as part of plume-tracking study were heavily confounded by non-point source contamination from onshore. These sources heavily confounded the interpretation of spatial trends everywhere, except for very close to the diffuser structure. At the time of the study, the influence of these other contamination sources, such as Morro Creek, was not widely recognized. As a result, many observations were misinterpreted as being discharge-related. Although the study contained no credible evidence for a direct connection between the discharge and the Estuary, shellfish contamination was incorrectly ascribed to the discharge. However, after disinfection of effluent was instituted, shellfish contamination continued within the Estuary unabated. Recent microbial source tracking studies using genetic fingerprinting have identified the principal origins of bacterial contamination within the Estuary as emanating from the discharge of Los Osos and Chorro Creeks.[23]

f) The MBCSD discharge does not tangibly combine with the discharges from "*at least five other NPDES-permitted sources*," as stated by NRDC.[24] The five sources cited by NRDC are too far from the MBCSD discharge to commingle in any material manner. The intermittent discharge of small volumes of treated groundwater (0.21 MGD) from Chevron Estero Marine Terminal are too distant (8,000 ft) from the MBCSD discharge to conceivable combine tangibly with the wastewater constituents within the MBCSD discharge, which are restricted to a distance of approximated 50 ft from the discharge location. The Morro Bay Desalination Plant and the Morro Bay Power Plant discharges, which are not currently discharging, consist purely of saltwater without contaminants. The other two discharges noted by NRDC are onshore surface water discharges.

3. **The MBCSD discharge does not pose a tangible human health risk.** The two-decade long record of intensive monitoring data demonstrates that the discharge has no perceptible impact on recreational activities within Estero Bay. The fact that all of the MBCSD effluent is thoroughly disinfected irrefutably eliminates the discharge as the potential source of elevated beach coliform. Moreover, the 28 quarterly water-quality surveys conducted during the current permit cycle have repeatedly demonstrated that perceptible concentrations of extremely dilute wastewater constituents are restricted to a highly localized region around the diffuser structure. Drogue drifter deployments demonstrated that not even very dilute quantities of wastewater constituents impinge on the shoreline. Compared to this quantitative evidence, contentions by NRDC[25] and Gold [26] regarding significant health risks are neither factually based nor accurate.

a) Discharged effluent does not degrade beach water quality. The MBCSD effluent is thoroughly disinfected and it cannot conceivably be considered to contribute to any perceived degradation in shoreline water quality. In point of fact, ninety-six percent of the time, the MBCSD effluent quality meets recreational water-contact standards[27] immediately upon discharge from the outfall. After accounting for the minimum initial dilution that occurs within 50 ft of the discharge, coliform densities measured in all effluent samples easily meet the more stringent bacterial standard imposed for shellfish

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[23] Kitts, C., M.A. Moline, A. Schaffner, M. Samadpour, K. McNeill, and S. Duffield. 2002. *Identifying the Sources of Escherichia coli Contamination to the Shellfish Growing Areas of the Morro Bay Estuary*, Final Report 3/29/2002.

[24] NRDC, at footnote 207 on page 40.

[25] NRDC, at 2, 39, 41, and 42.

[26] Gold, at 2, 3, and 4.

[27] Here, the instantaneous measurements of total coliform density within 3,318 effluent samples collected since 1993 were conservatively compared with the 30-day mean fecal coliform limit of 200 MPN/100 ml that applies to water-contact recreation per the COP.

harvesting areas.[28] Consequently, the discharge cannot possibly be tangibly contributing to the elevated coliform concentrations that are occasionally observed along the adjacent coastline, which is located more than 2,700 ft from the discharge.

The demonstrated effectiveness of the MBCSD disinfection process renders any further analysis moot with respect to assessments of potential discharge-induced impacts on human health. There is simply no nexus between absence of bacterial levels in the discharge and bacterial levels measured at the shoreline. Consequently, discussions concerning plume trajectories or variations in surfzone bacteria are irrelevant. Nevertheless, NRDC and Gold devote considerable attention to these topics in their comments, so they are addressed below.

b) **Rare instances of elevated beach coliform densities are associated with identifiable sources unrelated to the discharge.** The MBCSD surfzone monitoring program consists of frequent sampling within a dense station pattern along Morro Strand State Beach. The spatial and temporal sampling rate significantly exceeds that of the Heal-the-Bay report card cited by Gold.[29] The majority of the 3,500 MBCSD surfzone coliform measurements recorded over the current permit cycle reflect densities at or below the detection limit of 2 MPN/100 ml. A relatively small fraction of samples (about 10%) had coliform densities exceeding the shellfish harvesting standard. The vast majority of these elevated beach samples were associated with clearly identifiable non-point source contamination that was uncorrelated with periods of elevated effluent coliform levels. As described by extensive analysis of the data,[30] the elevated beach coliform events are typically associated with non-fecal coliform, rainfall events, measurements at distant reference stations, identifiable shoreline organic debris, or outflow from Morro Creek, which is recognized as a source of coliform contamination.

c) **Morro Creek was responsible for the brief period of poor beach report cards.** The most significant source of coliform contamination along Morro Strand State Beach is Morro Creek outflow. Its non-point source contributions are of particular interest insofar as explaining elevated shoreline coliform events cited by Gold in his beach report cards. His cursory review of the report cards and his unfamiliarity with the area led to his misguided speculation that "*…influences beyond storm water discharge…could include the Plant's effluent.*"[31] As discussed above, the vast monitoring record unequivocally demonstrates that effluent disinfection, dilution, and the discharge's offshore distance preclude any tangible wastewater contribution to shoreline bacteria. Gold bases his speculation on a review of only 78 grades[32] given in the past three years to a shoreline location at the projection of Atascadero Road. As evidence of the MBCSD's influence, he notes that this location received relatively poor report cards of surfzone bacterial levels in the 2004-2005 season compared to prior years. Although this sampling location is adjacent to the treatment plant, it is located 3,000 ft to the south of the portion of the shoreline adjacent to the offshore discharge point. Consequently, it is not as diagnostic of potential discharge-related impacts as surfzone samples collected as part of the MBCSD monitoring at a location directly shoreward of the discharge point.

In any regard, there is a simple explanation for the temporary reduction in water quality found in the Atascadero Road report cards during the winter months of 2005. Namely, during 2004, the mouth of Morro Creek migrated northward, and was directly adjacent to the sampling site in 2005 (Figure 1). All of the "F's" at this beach location occurred between 11 January 2005 and 28 March 2005, when the

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[28] The shellfish harvesting standard specifies that the median total coliform density shall not exceed 70 MPN/100 ml. This standard is met 92% of the time by the effluent prior to discharge and 100% of the time after initial dilution.

[29] Gold, at 2, 3, and 4.

[30] See for example, the 2005 Annual Monitoring Report, at Section 3.3.4.

[31] Gold, at 2.

[32] Compare this with the 3,500 surfzone coliform measurements analyzed by the MBCSD, USEPA, and RWQCB over the past seven years.

Figure 1. Aerial View of the Morro Creek Outflow Location

area was experiencing heavy rainfall from a series of storms that produced high creek flows. During that same period, effluent coliform levels never exceeded the water-contact bacterial standard and therefore, could not have contributed to the "failing" report-cards.

d) **The bacteriological monitoring data collected by the MBCSD, and the analysis and review by the USEPA and RWQCB, are current, comprehensive, and irrefutably demonstrate that the discharge poses no potential health risk.** As described above, the long record of consistently low bacterial densities in disinfected effluent demonstrates the absence of a discharge-related health risk along the shoreline, without even considering the equally extensive record of surfzone measurements. Both of these bacteriological data sets are submitted on a monthly basis to both the USEPA and RWQCB. An extensive analysis of these data sets was performed as recently as last month, as part of the 2005 Annual Report. In light of these facts, there is no basis for the NRDC and Gold statements that the data "*…are insufficient to support their conclusions that the Plant's discharge poses no potential health risk…*," that the analysis fails "*…to account for recent data that undercuts the fundamental conclusion that the Plant is not degrading beach water quality…*," or that the analysis does not "*…adequately evaluate current bacteria data at beaches adjacent to the outfall.*" Insofar as monitoring protocols, MBCSD surfzone samples are collected from the shoreline by hand, without wading or swimming out into the surfzone, so they are equivalent to a "*…sample at ankle depth*."[33]

e) **The discharge's depth and offshore distance provides substantial additional protection.** NRDC[34] and Gold[35] grossly distort the facts when they compare the MBCSD outfall "*…only half a mile offshore at a depth of less than 20 meters …*" with "*…**most** southern California outfalls [that] are 3 to 5 miles offshore in 60 meters depth…*" (emphasis added). In fact, the inventory of ocean dischargers cited by NRDC[36] show that few discharges are deeper or more distant than the MBCSD discharge, and that all of those dischargers produce a far greater volume of effluent. Of the seventeen ocean dischargers in southern California, only three have diffusers at depths of 60 m or greater. Each of these three have discharge volumes more than 100-times larger than that of the MBCSD discharge. Additionally, only

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[33] Gold, at 3.
[34] NRDC, at 41
[35] Gold, at 2 incorrectly states that "*…outfalls for most plants in southern California are much further* [sic] *offshore…and at least 60 meters in depth.*"
[36] NRDC, at Note 218: Heal the Ocean. 2005 Discharge Inventory for the State of California.

four of the seventeen southern California dischargers have diffusers located more than 3 miles from shore.

Furthermore, in discussing the discharge locations recommended by the World Health Organization (WHO),[37] NRDC is guilty of quote mining by failing to add the qualifying statement that "*In establishing safe depths and distance from shore, consideration must be given to local ocean conditions and **the amount of sewage discharged***" (emphasis added). Thus, a more relevant and accurate ranking of the MBCSD discharge shows that it is located farther offshore, and is the second deepest of the sixteen California discharges with volumes less than 4.8 MGD. Finally, the WHO assessment factors[38] demonstrate that the MBCSD discharge has a very low potential human health risk for exposure to sewage because of 1) an effective outfall, 2) disinfection of all effluent, 3) secondary treatment of nearly all of the effluent, and 4) the low discharge volume arising from a small population.

f) **Quarterly assessments of plume dispersion and transport repeatedly demonstrate that measurable concentrations of wastewater constituents are restricted to a highly localized area around the diffuser structure, and do not impinge on the shoreline.** Based on his extraordinarily cursory review, Gold contends that there is no "*…monitoring information that would allow them to determine if the Plant's effluent plume comes back to shore…, [in] particular, …[a] recent plume tracking study…that accounts for varying conditions season, current, swell height, and temperature….There is no indication that monitoring occurs under varying tidal conditions.*"[39] Unfortunately, Gold either was not given,[40] or choose not to read the public monitoring record covering the current permit cycle. Otherwise, he would have known that the MBCSD regularly conducts plume-tracking studies as part of their quarterly water-quality surveys.

In fact, detailed plume delineation studies have been conducted on 28 separate occasions during the past seven years. As documented in the quarterly water-quality surveys, the annual reports, and the supplement, these studies encompass the full range of current, wave height, tide, temperature (stratification), and seasonal conditions. The studies included trajectory analyses of satellite-tracked drogued drifters that are released at the center of the diffuser structure. In all 28 surveys, highly sensitive instrumentation detected, delineated, and tracked extremely dilute wastewater constituents surrounding the diffuser structure. All of these measurements demonstrate that perceptible discharge-related water-quality anomalies are largely restricted to a distance of 50 ft from the diffuser. There has never been any indication that the plume impinged on the adjacent shoreline, which is more than 2,700 ft from the discharge point. On the contrary, current velocities determined from drifter trajectories as summarized in the supplement[41] demonstrate that flow was preferentially directed along-shore. Current velocity is directed toward shore only 6% of the time, and then, only for a brief period with very low speed.

g) **Mussel monitoring has demonstrated an absence of a wide range of bacterial pathogens within the MBCSD discharge.** Contrary to assertions by Gold,[42] the MBCSD has analyzed bacterial tissue burdens in both indigenous[43] and outplanted[44] mussels. Resident mussels collected upcoast and downcoast of the discharge, as well as at distant reference stations, were analyzed for fecal coliform bacteria by the San Luis Obispo Health Department Laboratory. There were a few instances when

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37 Heal the Ocean. 2005. Discharge Inventory for the State of California, at 4.

38 Ib. at Appendix C: Potential human health risks arising from exposure to sewage WHO, 2000.

39 Gold, at 3.

40 As described at 1.b), NRDC was provided with a copy of the 2004 Annual Monitoring Report.

41 Supplement, at Figure 7.

42 Gold, at 3: "*…no data was provided on bacterial densities in mussels…mussels should have been analyzed for fecal indicators.*"

43 MRS. 1995. Offshore Monitoring and Reporting Program, 1994 Annual Report. Submitted February 1995 to the MBCSD, at Section 3.2.10 Coliform Bacteria - Shellfish Monitoring.

44 2004 Annual Report, at Section 2.2.11.

tissue burdens exceeded acceptance levels for commercial sale. These samples were collected shortly after rainfall events, and the control samples collected at the reference sites also contained elevated densities on these occasions. Conversely, effluent coliform densities were uniformly low for an extended period prior to when the mussels were collected with elevated tissue burdens.

Caged mussels were also deployed on the outfall buoys marking the MBCSD diffuser structure during the early dry season, late dry season, early wet season, and late wet season of 2003 and 2004. The mussels were allowed to filter-feed for a minimum of 30 days after outplanting. Thirty mussels were retrieved from each deployment and tests were negative for a wide suite of bacterial pathogens including: *Campylobacter, Clostridium perfringens, Plesiomonas shigelloides, Salmonella, and Vibrio* spp. (*cholerae, parahaemolyticus*, etc.). These quantitative observations strongly contradict speculation by NRDC and CEA that suspended solids within the MBCSD wastestream "…*interfere with disinfection by shielding pathogenic organisms from the disinfectant*."[45]

4. **Monitoring data demonstrate the ability of the MBCSD discharge to comply with water-quality objectives.** Despite repeated assertions to the contrary by NRDC and CEA, the massive amount of monitoring data collected over the past two decades demonstrates that the MBCSD discharge has consistently met water-quality standards. Statistical projections demonstrate that it will continue to do so for the next two permit terms. Furthermore, the intensive monitoring program is more than capable of detecting any potentially significant impacts to water quality should they arise during that time.

   a) **There is no evidence that discharge-related impacts to the marine environment will occur if the MBCSD continues to operate under a 301(h) modification.** There is no scientific basis for the CEA statement that "*Providing less than secondary treatment results in adverse water-quality impacts when compared with full secondary treatment*." It is not true of discharges in the open-ocean environment in general, and is particularly untrue of the low-volume MBCSD discharge, where solids concentrations are already close to secondary standards. Discharge limitations derived from secondary treatment standards are technology based, and do not always reflect the potential for environmental impacts. Simply because a technology is capable of achieving a particular removal rate does not mean that imposing a technology-based requirement would be more protective of the environment. In fact, Section 301(h) of the Clean Water Act (CWA) was first proposed largely because of scientific testimony[46] that challenged the environmental efficacy of imposing purely technology-based limits on open-ocean dischargers. Many studies conducted in the decades since the first Section 301(h)-modified permits were issued, have confirmed the veracity of the original testimony.[47]

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45 CEA, at 2 and 3, and NRDC, at 1 and 45.

46 Isaacs, J.D. 1978. *Testimony of Prof. John D. Isaacs, Director, Institute of Marine Resources, University of California, San Diego.* Pp 36-44 In: Modification of secondary treatment requirements for discharges into marine waters. Hearings before the Subcommittee on Water Resources of the Committee on Public Works and Transportation, House of Representatives, Ninety-fifth Congress, Second Session, May 24 and 25, 1978. U.S. Government Printing Office, Washington D.C.

47 For example: Mearns, A.J. 1980. *Effects of municipal discharges on open coastal ecosystems*. Marine Environmental Pollution, 2:25-66; National Academy of Sciences. 1993. *Managing Wastewater in Coastal Urban Areas.* National Research Council; Tegner et al. 1995. *Effects of a sewage spill on a kelp forest community: catastrophe or disturbance?* Marine Environmental Research, 40(2), 181-224; Diener et al. 1995. *Infaunal patterns in the vicinity of a small coastal wastewater outfall and the lack of infaunal community response to secondary treatment.* Bulletin Southern California Academy of Sciences. 94(1):5-20; Diener, et al. 1995. *Spatial and temporal patterns of the infaunal community near a major ocean outfall in southern California.* Marine Pollution Bulletin. 30:861-878; Dorsey et al. 1995. *Changes in assemblages of infaunal organisms around wastewater outfalls in Santa Monica Bay, California.* Bulletin of the Southern California Academy of Sciences, 94(1), 5-20; Girgg, et al. 1995. *Environmental protection misapplied: alleged versus documented impacts of a deep ocean sewage outfall in Hawaii*. Ambio 24(2), 125-128; Loehr, L.C. 1986. *The exclusion of science in major treatment decisions in the state of Washington.* Marine Pollution Bulletin, 17(11);

Given this scientific documentation, it is disingenuous for NRDC and CEA to suggest that the MBCSD discharge cannot meet water-quality standards because a full-secondary treatment requirement is not immediately imposed on the discharge. This is especially true considering that there is no tangible direct evidence that the current discharge is impairing the environment based on a monitoring program "*…among the most comprehensive of all municipal ocean discharges of less than 5 MGD in California*."[5] Water-quality measurements collected during quarterly water-quality surveys repeatedly demonstrate that the energetic marine environment within Estero Bay rapidly and easily assimilates the MBCSD particulates within a few meters of the discharge.

b) **The discharge has and will continue to comply with the requirements of a 301(h)-modified permit.** Based on an exhaustive independent review of the vast array of monitoring data, the USEPA determined that the MBCSD discharge met each of the nine criteria set forth in Section 301(h) of the CWA. Based on their thorough evaluation, the EPA regional administrator tentatively decided that the MBCSD should be granted a discharge permit that modifies the TSS and BOD requirements as provided in the Federal Regulations. This determination was made not once, but on two prior occasions, and the RWQCB and the California Coastal Commission concurred that the discharge complies with the State of California's water-quality standards each time. Their conclusions were based on the long and consistent record of compliance associated with the MBCSD discharge. An unforeseen failure in a mechanical component occasionally occurs that results in a brief exception to permit limits, but those occasions are brief and extremely rare. During the current permit cycle, exceptions to the permit generally constitute less than 0.5% of the samples collected.

Nevertheless, based on these rare, brief, out-of-compliance events, NRDC speculates that the MBCSD "*…cannot demonstrate that the modified discharge will meet the requirements for compliance with water quality standards*."[48] This is simply not true. Not only does the discharge regularly and consistently meet water quality standards, but there is no evidence of any decline in the effectiveness of the treatment process over the two decades of well-documented operational history. In fact, treatment performance in 2005 was among the highest on record, and out of the thousands of measurements collected that year, there was not a single exception to permit limits.[49]

c) **The rare exceptions to permit limitations have been brief and were attributable to identifiable causes unrelated to the overall efficacy of the treatment process.** NRDC emphasizes a few isolated exceptions to effluent limitations that have occurred in the long record of MBCSD monitoring. The origins of these rare exceptions are well documented. Their isolated nature and relationship to external events clearly indicate that they were unrelated to the overall ability of the treatment process to "*…comply with specific water quality standards*."[50]

TSS: For example, NRDC cites three TSS measurements that slightly exceeded an instantaneous limit over a brief period of time. All three exceedances were related to the same isolated event. Considering that there were 2,200 other TSS measurements that were all in compliance during the current permit period, these three elevated levels hardly seem to represent "*…a record of TSS violations…that clearly show the facility has not consistently met applicable water quality standards*" as claimed by NRDC.[51]

Coliform: Similarly, NRDC cites two brief occasions when temporary malfunctions in pumps feeding the chlorine auto analyzer resulted in elevated median effluent coliform densities. Potential future

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Loehr and Brooks. 1995. *Judge scraps EPA-ordered sewage requirements for San Diego.* Marine Pollution Bulletin, 30(5), 354-355; Chapman, et al. 1996. *A triad study of sediment quality associated wit a major, relatively untreated marine sewage discharge.* Marine Pollution Bulletin 32(1), 47-67; and Chen, et al. 1974. *Trace metals in wastewater effluents.* Journal Water Pollution Control Federation, 46(12), 2663-2678.

48 NRDC at 39, Part 3C

49 2005, at the Executive Summary

50 NRDC, at Part 3C, Section C.

51 NRDC, at 42 and 43.

instances of elevated effluent coliform were eliminated after sampling pumps were rebuilt and alarm systems were installed. NRDC grossly mischaracterizes these two brief mechanical problems as a "*…malfunctioning, inadequate treatment process…* [that is] *…incapable of consistently disinfecting the effluent to meet the limits*."[52] This is an unreasonable assertion considering that 2,900 other measurements collected through 2005 were in strict compliance with the limits. Furthermore, NRDC's section heading incorrectly implies that there were violations of effluent limits related to other pathogens, but they fail to specify what those violations were. This is because there were none.[53]

Chlorine: NRDC also cites a few instances of elevated effluent chlorine that were also caused by initial difficulties with the sampling device that paces the chlorination/dechlorination process. The sample-supply line was subsequently redesigned to improve flow and a more rigorous maintenance program was initiated. These changes eliminated the sampling problem and the related chlorine violations have not occurred since. Nevertheless, NRDC claims that the mechanical problems were somehow related to the increased presence of solids in the chlorine contact chamber and that "*Given these violations, EPA and the Regional Board staff's finding that the Plant is in compliance with the chlorine residual limitation is unsupported.*"[54] However, this claim completely ignores the 2,900 other chlorine measurements that were found to be in compliance during the course of the current permit.

Dioxin: NRDC[55] goes on to incorrectly state that "*…the Plant's data reveals a series of violations…*" for dioxin, when in fact, only one highly suspect measurement of dioxin exceeded the permit limit. The July-2002 measurement in question exceeded the permit limit by an extremely small amount that was not reliably quantified. Few laboratories "*…can perform these analyses at the very low levels of detection necessary to determine if these compounds are present at concentrations of potential health significance.*"[56] Laboratory contamination at these extremely small concentrations is commonplace and adds great uncertainty to the reported results. In fact, the initial analysis of the July-2002 effluent sample erroneously reported a much higher dioxin concentration due to high dioxin concentrations in the certified "pure" water used to dilute the effluent sample.

In addition, contrary to the statements by NRDC and CEA, the presence of detectable dioxin concentrations in other effluent samples is irrelevant to an evaluation of plant compliance and performance. Dioxin is regularly detected in effluent samples because it is pervasive in the environment through deposition of air emissions onto the soil surface and subsequent erosion and run-off.[57] Only a few of the least toxic and most common of the 210 dioxin congeners are typically found in effluent samples. Their presence at very low, but detectable levels in the effluent is expected, and they are not indicative of "*…a more systematic problem*" as stated by CEA.[58] CEA does not provide any quantitative support or scientific basis for his assertions that dioxin "*…is readily adsorbed onto TSS...*," its "*…source…is the commercial laundry…*," or that "*…the current treatment…results in higher discharges….*" The latter unfounded claim is restated by NRDC[59] "*…that the Plant's current discharge of blended effluent results in higher discharges of dioxin than would effluent that was fully treated.*" How can this be true when the current discharge effectively meets secondary treatment standards for TSS the vast majority of time? The answer is that this unfounded assertion is simply not true.

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52 NRDC, at 43.
53 NRDC, at 3C.C.2. "*Violations of Total Coliform and Other Pathogens*"
54 NRDC, at 46.
55 NRDC, at 47.
56 State Water Resources Control Board (SWRCB). 1999. California Ocean Plan 1999-2002 Triennial Review Workplan. [http://www.swrcb.ca.gov/plnspols/oplans/9902atrwrkp/sectc.doc], at C-27
57 USEPA. 2004. NAS Review Draft of EPA's Exposure and Human Health Reassessment of 2,3,7,8-Tetrachlorodibenzo-p-Dioxin (TCDD) and Related Compounds.
58 CEA, at 3.
59 NRDC, at 47.

Metals: NRDC[59] makes the same incorrect statement for trace metals, namely, that "…*the Plant's data also shows a series of violations.*" The only difference in their misstatement for metals is that there has never been even one exception to the permit limitation on metals. In fact, out of the 325 metal analyses reported since 1993, there has never been a single concentration that even approached the permit limit. Again, the presence of "…*detectable levels…*" of metals in the effluent samples is meaningless because they are naturally occurring substances in the environment. Consequently, detectable concentrations are expected to routinely occur in effluent samples, and their presence does not represent a contaminant source as implied by NRDC. Finally, as described in 4.f) below, NRDC completely mischaracterize data on bulk-metal concentrations within Estero Bay sediments as showing "…*that levels of chromium, nickel, copper, and arsenic will likely accumulate in the near future above levels considered harmful to biota.*" The concentrations of these metals in bulk sediments are naturally high in the region but are unrelated to the discharge. Moreover, dissolved concentrations are not significantly elevated, so the bulk concentrations have little potential for biological impact regardless of how they compare with effects thresholds.

d) **Statistical analyses demonstrate that there is a high potential for future compliance with water-quality limits on chemical constituents.** Quantitative analyses of a decade of MBCSD effluent measurements have definitively demonstrated that the MBCSD discharge has a uniformly high compliance potential.[60] This Reasonable Potential Analysis (RPA) used the methodology currently recommended by the SWRCB for determining the need for effluent limitations in NPDES discharge permits. The RPA provides a conservative quantitative basis for evaluating each contaminant's potential for non-compliance with the COP. The RPA demonstrates that the potential for compliance is high for 60 of the 73 chemical constituents currently monitored on a semi-annual basis. The other thirteen compounds have never been detected in effluent samples except for an isolated measurement of DDT that was due to laboratory error. The RPA demonstrates that NRDC[61] incorrectly claims that "*the Draft Permit fails to meet the conditions set forth in 40 C.F.R. § 122.44(d) for all pollutants that…have a reasonable potential to cause… a violation of any State water quality standard.*"

e) **The well-designed monitoring program is capable of detecting minute changes in the marine environment.** The MBCSD ocean monitoring program is specifically designed to provide early warning of impairment to marine habitat or recreational use. It is among the most expensive and comprehensive monitoring programs for any discharger of its size in California. The monitoring protocols are continuously refined and the sampling design has a demonstrably high statistical power to detect changes in the marine environment.[11] The monitoring program has been conducted for two decades, allowing ample time for any potential impact from the discharge to be revealed. Since its inception, hundreds of thousands of measurements have been collected and analyzed to assess water and sediment quality surrounding the outfall, and on the adjacent shoreline. This vast array of measurements unequivocally demonstrates that recreational uses and habitat are not adversely affected by the MBCSD discharge. The monitoring requirements were developed in coordination with USEPA and RWQCB staff to specifically comply with federal 301(h) regulations and COP requirements. The sampling, analytical, and statistical techniques are commonly used in ocean monitoring programs throughout the nation, and they are regularly used in southern California to monitor the effects of large municipal dischargers on the ocean's ecosystems.

f) **There is no quantitative evidence that metals are accumulating around the outfall and affecting marine organisms.** CEA incorrectly implies that because "*Metals have been measured in the sediments near the outfall at levels exceeding the ERL and ERM for nickel and approaching the ERL*

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[60] MBCSD. 2004. Letter from Mr. Bruce Keogh, Wastewater Division Manager, MBCSD to Mr. Matt Thompson, RWQCB. Subject: *Submittal of quantitative documentation in support of reductions in the monitoring frequency for chemical contaminants with the MBCSD effluent*. 19 July.

[61] NRDC, at 40.

for chromium…[that] *"10% of the benthic organisms are expected to be adversely affected* [by chromium, and]*…50%…are expected to be adversely affected* [by nickel]*."*[62]

First, CEA exhibits an incredible statistical naiveté in stating that the ERL/ERM benchmarks reflect the percentage *"…of the benthic organisms…expected to be adversely affected*." This is analogous to flagrantly misinterpreting a 10% chance of rain to mean that 10% of the area will receive rain, when the correct interpretation is that rain occurs 10% of the time when similar atmospheric conditions have prevailed in the past. This may seem like a subtle distinction, but it is a fundamental precept for the correct interpretation of statistical probabilities.

Second, the concentrations of these two metals are naturally elevated in the mineralogy throughout the central coast, not just "*near the outfall*." Naturally occurring chromite is introduced into Central Coast waters by fluvial transport of eroded ultrabasic minerals found in the Franciscan formation. The Franciscan formation outcrops along the headlands north of Point Estero and in the Santa Ynez Mountains. The concentrations of these two metals within sediments bear no relation to the discharge. There is no concentration gradient related to outfall proximity so there is no "*…accumulation of toxic metals around its discharge pipe…,*"[63] and the mass emissions of these metals in the MBCSD discharge are too small to account for the observed sediment concentrations surrounding the outfall.[64] In fact, nickel has never been detected in the MBCSD effluent, and when chromium has occasionally been detected, it has been at concentrations more than 45 times lower than the permit limit.[65] Consequently, "*Implementation of full secondary treatment...*" could not possibly result in any tangible reduction in the Plant's *"...contribution to the nickel and chromium accumulation in the sediments near the outfall."*[66]

Third, the biological effects thresholds (ERL/ERM) for these two metals are highly uncertain compared to thresholds determined for other contaminants. This uncertainty is documented by the NOAA investigators[67] who established the thresholds, and is not the "*…the argument of the …Plant…*," as asserted by CEA. For example, the developers of the thresholds state that the incidence of effects in the toxicological studies used to establish the levels for chromium were "*…greatly influenced and exaggerated by data from multiple tests conducted in only two field surveys*." Similarly, nickel concentrations exhibit a very weak relationship to the incidence of effects in the database used to establish the nickel threshold. The uncertainty in the thresholds of biological effects for these two metals arises because their bioavailability varies greatly for a given bulk sediment concentration.

Fourth and last, analysis of porewater samples within Estero Bay demonstrates that the naturally occurring elevated chromium and nickel concentrations in bulk sediment samples arise from metal that is bound into the mineral matrix of sand grains, where it has little influence on indigenous marine organisms.[68] Dissolved chromium concentrations within Estero Bay porewaters were comparable to

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62 CEA, at 3, and particularly, Footnote 4.

63 NRDC, at 1.

64 2005 Annual Report, at Section 4.2.2.

65 Supplement, at Table 3. This is a very conservative comparison between the highest measured total chromium concentration and the six-month median concentration for hexavalent chromium. According to CEA, at 4, "*Any hexavalent chromium…would be rapidly reduced to trivalent chromium*," so a more reasonable comparison would be with the limit on trivalent chromium, which is 4 million times higher than the highest measured total chromium concentration.

66 CEA, at 3.

67 Long, E.R., and L.G. Morgan. 1991. The potential for biological effects of sediment-sorbed contaminants tested in the National Status and Trends Program. U.S. Dept. Commerce, Nat. Oceanic Atmos. Admin., Off. Oceanogr. Mar. Assess., Rockville, MD. NOAA Tech. Mem. NOS OMA 52 175 pp. plus appendices; and Long, E.R., D.D. MacDonald, S.L. Smith and F.D. Calder. 1995. Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. Environmental Management 19(1):81-97.

68 2005 Annual Report, at 4-33 and 4-34, Section 4.2.3.

background concentrations in clean coastal seawater from other areas. Both dissolved chromium and nickel concentrations were far below the most-stringent saltwater criteria promulgated by the USEPA.

g) **The rationale for chronic toxicity testing in lieu of acute bioassays is sound.** The California State Water Resources Control Board[69] and the USEPA[70] have thoroughly examined this issue and have found that chronic testing is more appropriate for discharges with dilutions factors close to 100:1.[71] They further state that "*Where other factors are equal, chronic testing may be preferable since interim results taken from chronic test provide data on acute toxicity as well.*" This SWRCB finding directly contradicts the CEA statement "*…that acute and chronic toxicity testing measure different things*." [72] Even more disconcerting is the blatant CEA statement that the "*…Plant argues it should not have to do acute toxicity testing because it might fail the toxicity test*…." This is patently false. No such statement was ever made or implied.

NRDC completely misinterprets its own consultant concerning "*…acute toxicity caused by chlorine…*,"[73] when the entire CEA discussion[74] of chlorine pertains to its disinfection efficacy, not its toxicity within effluent. They further cite "*…other unambiguous violations of applicable standards*…," as evidence that the discharge cannot satisfy 301(h) standards, but fail to elaborate and specify what those might be.

5. **Limited increases in population over the next decade will not tangibly affect the MBCSD's ability to comply with discharge requirements on TSS and BOD.** NRDC's statement[75] that "*…the Plant is highly likely to process additional volumes of effluent in the next five years, a fact which will exacerbate…the rate of effective disinfection and water quality standards compliance*" is neither factually based nor pertinent. In a similar vein, the Sierra Club states that the discharge permit should be denied because "*The volume of wastewater treated by the plant is sure to increase over the next five years,*" although they do not explain why this is a reason to deny the permit. Conversely, Section 301(h) does not explicitly restrict flow rates or the fraction of primary treated effluent. Instead, it limits increases in BOD and TSS concentrations and loadings above the discharge specified in the permit. In that regard, it is noteworthy that when the current discharge permit was issued, the allowed limit on TSS and BOD mass-emissions was reduced 12.7%. Nevertheless, TSS and BOD concentrations and emissions are projected to remain well within these more-restrictive limits throughout at least the next two permit cycles.

a) **The vast majority of time, the entirety of the MBCSD flow receives secondary treatment.** In light of this fact, the NRDC statement[76] that the "*…flow rate is currently at levels that allow some of the flow to be treated to secondary treatment levels…*," is misleading because it downplays the fraction of flow that actually receives secondary treatment. In fact, effluent usually consists entirely of wastewater that has received secondary treatment. Blending of primary- and secondary-treated wastewater is only required during periods of particularly high flow, such as during rainstorms and some holidays. During 2005, the blending valve was open only 7.5% of the time.[77] Optimizing the treatment process in this manner has resulted in steadily increasing solids removal rates, from 86% in 2002 to 93.3% in 2005.[78]

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[69] SWRCB. 2000. Division of Water Quality. Draft Final Functional Equivalent Document Amendment of the Water Quality Control Plan for Ocean Waters of California. COP. September 1, at 14.
[70] USEPA 1991. Technical Support Document for Water Quality-based Toxics Control. USEPA, Office of Water. EPA/505/2-90-001.
[71] The MBCSD minimum initial dilution has been determined to be 133:1.
[72] CEA, at 4.
[73] NRDC, at 1.
[74] CEA, at 2 and 3.
[75] NRDC, at 6.
[76] NRDC, at 43.
[77] 2005 Annual Report, at 2-1.
[78] 2005 Annual Report, at Table 5.1 on page 5-2.

These quantitative measures of plant performance negate the NRDC[79] statement that the "…*Plant's assertion that removal rates will remain stable over the next five years is inaccurate*." In fact, TSS removal rates have actually markedly improved in the three years since the "*Plant's assertion*" was made using 2002 data.

b) **Previous projections of increases in flow, BOD, and TSS represent upper-bound estimates.** The TSD analysis clearly stated that the projected increases over upcoming permit cycles were upper-bound estimates because a number of conservative assumptions were included in the analysis. For example, projected flows were not adjusted for the observed decline in per-capita water usage. Additionally, MBCSD is actively identifying and eliminating sources of groundwater infiltration into the collection system. Based on the flow-source analysis conducted in 2005, infiltration contributes at least 0.2 MGD to the overall flow.[80] Even eliminating a modest amount of infiltration could easily offset projected annual flow increases (0.01 MGD) related to population growth. This will forestall projected increases in the proportion of flow that would need to be bypassed around the secondary treatment process, and stable BOD and TSS emissions could be maintained for many years as a result.

c) **There is no evidence of a decline in plant performance.** On the contrary, the most recent monitoring documented exceptionally high operational performance by the treatment plant. Over the two decades of modern plant operations, all the major effluent constituents, including TSS, BOD, and oil and grease (O&G) have had consistently lower concentrations and mass emissions than the permitted maximums. However, in the last three years, the annual averages of several key diagnostics of treatment performance were some of the best reported in nearly two decades of monitoring. In 2005, the suspended solids concentration within effluent was the second lowest on record, while the solids removal rate was the highest on record. Combined with the low total flows, the plant's solids-removal process resulted in the lowest annual mass emission of suspended solids recorded since monitoring began in 1986. Similarly, the average effluent O&G and BOD concentrations in 2005 were much lower than average, despite higher-than-average influent concentrations.

6. **There is no plausible link between the MBCSD discharge and the occurrence of *T. gondii* seropositivity in otters.** The NRDC repeatedly implies that the MBCSD discharge is somehow responsible for the higher occurrence of *T. gondii* seropositivity found in otters in the Estero Bay region as compared with other areas of the coastline. The NRDC hinges its entire case on the fact that the highly localized MBCSD discharge happens to fall within a large coastline segment containing elevated *T. gondii* seropositivity. Not only is it inappropriate to ascribe causality to geographic coincidence, but it ignores the substantial scientific evidence indicating that there is no plausible nexus between the discharge, the otters, and the presence of *T. gondii* oocysts.

a) **The MBCSD discharge is too limited to contribute to the observed *T. gondii* seropositivity in otters, and thus the spatial correlation is weak.** The NRDC[81] provides an extremely misleading aerial photograph showing the geographic center of the region of elevated *T. gondii* seropositivity, a region which spans all of Estero Bay. The geographic center of this large region happens to be in close proximity to the outfall. This spatial comparison is meaningless because of the large difference in the areas associated with these occurrences. The actual Estero-Bay area of elevated *T. gondii* seropositivity, on which NRDC base their claims, extends a total of 20 km extending from Caycos south to Hazard Canyon. This area is huge (30 km^2) when huge compared to the footprint of the MBCSD discharge (0.003 km^2).[82] Given the large difference in areas, there is a very low likelihood that otters within Estero Bay would randomly encounter the effluent plume. In addition, there is no evidence that otters

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79 NRDC, at 45, citing a statement in the TSD
80 2005 Annual Report, Table 2.6 at Page 2-24.
81 NRDC, at 35. The area of elevated *T. gondii* seropositivity within Estero Bay is 1.5 km by 20 km.
82 2005 Annual Report, at 3-15.

preferentially forage near or remain in the vicinity of the outfall for extended periods.[83] Thus, even if the discharge contained high concentrations of *T. gondii* oocysts, the likelihood of an otter becoming infected would be extremely unlikely.

b) **Wastewater discharge is not a plausible source of *T. gondii* oocysts**. The seminal otter toxoplasmosis study[84] found strong evidence that the occurrence of *T. gondii* was highly correlated with non-point source freshwater flow, namely, rivers and streams, and not with proximity to sewage outfalls. Specifically, "*seropositivity to T. gondii was not significantly associated with human population density (P=.293), or proximity to sewage outfalls (P=0.955) but was highly correlated with freshwater flow (P<0.001)*." The size of the associated error probabilities, or "*P*-values," demonstrates an unusually high degree of confidence in the correlation with freshwater outflow, and a corresponding lack of correlation with wastewater discharge. Conrad et al (2005) concur with this assessment when they state that "*the most likely source of infection is by infectious, environmentally resistant oocysts that are shed in the feces of felids and transported via freshwater runoff into the marine ecosystem.*"[85]

Clearly, the freshwater flow pathway implicated in recent studies is a plausible source for *T. gondii* input to the Morro Bay Estuary considering the number and feral nature of feline carriers likely to reside within the 48,000 acre watershed that feeds the estuary. In contrast, the MBCSD wastewater is not a plausible source, especially considering the life history of *T. gondii* oocysts.[86] Viable *T. gondii* organisms are excreted in cat feces for only two to three weeks, and most healthy cats shed oocysts only during a single, brief, acute infection stage. A survey of cats at humane shelters demonstrated that 50% of the cats were infected with toxoplasmosis, but of these, only about 1% were actually passing oocysts. These oocysts only remain infective for two to three days. There was also some speculation that disposal of used cat litter into the collection system could represent a source of the oocysts. However, this pathway is extremely unlikely given that the introduction of cat feces contaminated with *T. gondii* oocysts into the collection system would be extraordinarily rare and highly intermittent

c) **Mussel tissue tests documented a general absence of *T. gondii* within the MBCSD discharge.** Thirty caged mussels deployed near the MBCSD diffuser structure on four separate occasions tested negative for the presence of *T. gondii* oocysts. Measuring *T. gondii* concentrations in mussel tissue is a particularly appropriate test because these filter-feeding shellfish amplify and integrate intermittent contaminant input over time, and because ingestion of mussel tissue by otters is one of the pathways identified for *T. gondii* exposure. Repeated sampling over a one-year period provided a fairly complete exposure to MBCSD effluent. NRDC[87] attempts to downplay the significance of these results simply because the assay had a detection limit. All valid quantitative tests, especially measurements of chemical concentrations, have a detection limit. This does not mean that these tests "*…are limited and the study is incomplete.*" Also, it was not a "*single*" study as repeatedly characterized by the NRDC. It was a comprehensive survey that included four separate long-duration deployments covering all seasons within a one-year period. These site-specific quantitative results, which show a general absence of detectable oocyst densities in the MBCSD discharge, combined with the small MBCSD discharge footprint, the regional lack of correlation between wastewater and seropositivity, and the low likelihood

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83 Beneficial-use observations recorded as part of the MBCSD water-quality monitoring surveys occasionally note the presence of otters during transit from Morro Bay, and within the survey region near the outfall, but otter observations at the outfall are no more common than in other parts of Estero Bay, and far less common than otter observations within the Morro Bay Estuary.

84 Miller et al. 2002. Coastal freshwater runoff is a risk factor for *Toxoplasma gondii* infection of southern sea otters (*Enhydra lutris nereis*), 32 International Journal for Parasitology 997, 997-98.

85 Conrad et al. 2005. *Transmission of Toxoplasma: Clues from the study of sea otters as sentinels of Toxoplasma gondii flow into the marine environment.* International Journal for Parasitology 35(2005) 1155-1168. at 1155.

86 http://www.biosci.ohio-state.edu/~parasite/more_toxoplasma.html

87 NRDC, at 12.

of oocyst input into the collection system, rule out the MBCSD discharge as a potential source for elevated *Toxoplasmosis* in otters near Estero Bay.

7. **There is no quantitative evidence supporting the claim that the area around the MBCSD discharge lacks a balanced indigenous marine population (BIP).** The massive amount of MBCSD monitoring data and associated analyses demonstrates that a healthy BIP flourishes around the discharge. In contrast, NRDC provides no substantive quantitative evidence that the marine population near the outfall is not in balance, even after accounting for the regional dynamics of otter populations.

   a) **A balanced population of sensitive filter-feeding organisms live in clean sediments surrounding the outfall.** Infauna living within the sediments surrounding the outfall are highly diagnostic of the status of indigenous populations as a whole, and are sensitive indicators of marine pollution. They have limited mobility and cannot easily escape exposure to contaminants in their immediate environment. They have well-defined responses to contaminant exposure and are located close to the seafloor source of the discharge. They are a major food source for more-mobile epifaunal and pelagic marine organisms, such as crabs, finfish, and marine mammals. Finally, many infauna are filter feeders that bioaccumulate contaminants when standard chemical assays of water samples are unable to detect low-level concentrations. Over 142,000 infaunal specimens have been collected near the outfall during the two-decade monitoring program. Highly sensitive spatiotemporal analysis of these data exhibits no evidence of benthic degradation and quantitatively demonstrates that a healthy indigenous infaunal community with a uniformly high diversity resides near the outfall.[88]

   b) **The southern sea otter population is not in a state of decline.** The repetitious assertions by NRDC[89] grossly misrepresent otter demographics by claiming that "*The southern sea otter...is a threatened marine mammal species whose population is in decline*", and "*Recently… the sea otter has suffered a steady and grave decline.*" Although the population has, indeed, undergone several periods of decline on its road to recovery,[90] since 1999, the southern sea otter population has grown substantially, and it is not currently considered to be in decline (Figure 2).[91] Following the conclusion of the spring 2004 otter census, U.S. Geological Survey (USGS) biologist Brian Hatfield confirmed that the southern sea otter "*is now in a positive growth trend.*"[92] In fact, during 2004, the otter population reached a record high of 2,825 individuals, while the spring 2005 survey, led by scientists at the USGS, tallied a total of 2,735 otters.[93] Although the 2005 data show a 3.2% decrease from the 2004 high, the population remains at the second highest level on record since the inception of quantitative semi-annual surveys in 1983.

      Additionally, NRDC's observation that the current statewide population *"has not grown significantly since 1994,"*[94] purposefully omits the decline that they previously were so adamant to draw attention to,[95] in order to minimize the true extent of the population growth that has subsequently occurred. Although the statewide otter population has increased approximately 14% since 1994, it has increased by more than double that (30%) since 1999, when the population had shrunk to a low of 2090 individuals.[96]

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88 2005 Annual Report, at Chapter 4.

89 NRDC, at 6,8, 19 (footnote), 26-27, 34, and 56.

90 Documented declines in the statewide otter population occurred from the late 1970s to the early 1980s, as well as from 1995-1999.

91 Brian Hatfield, USGS biologist with WERC, personal communication concerning current status of southern sea otter population, 17 February 2006.

92 The Otter Project: USGS Otter Survey Results, Spring 2004, Accessed online, February 2006, at: http://www.otterproject.org/site/pp.asp?c=8pIKIYMIG&b=28114

93 The Otter Project: USGS Otter Survey Results, Spring 2004, 2005.

94 NRDC, at 8, 27.

95 The decline in question, from 1995-1999, is characterized as a "modest decline" in the Final Revised Recovery Plan Recovery Plan, (at D-10).

96 The Otter Project. USGS Spring and Fall Census Reports for 1997-2005.

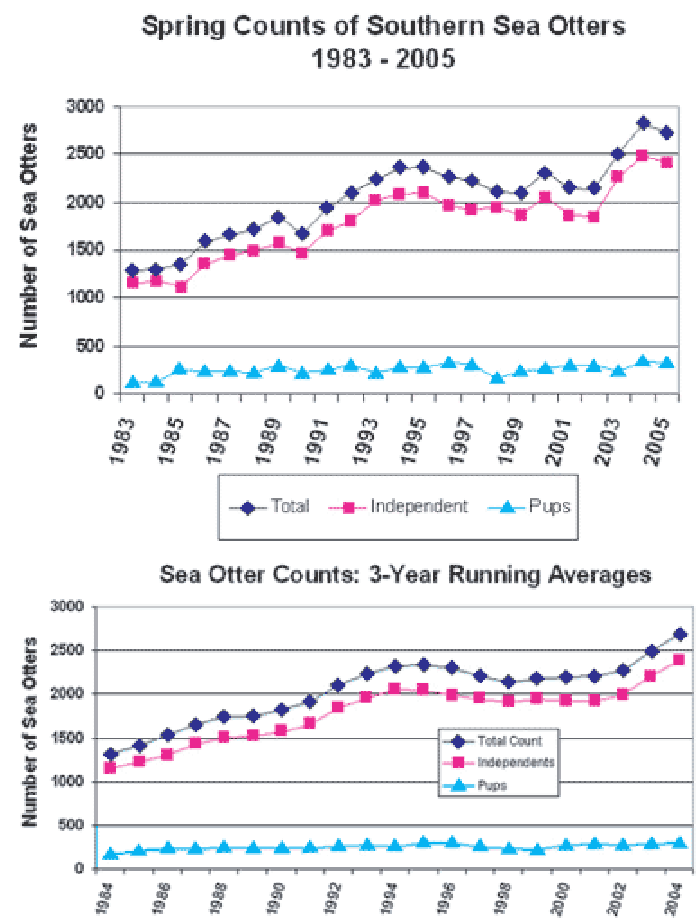

Figure 2. Otter Census Data

Finally, as it is recognized that little insight into overall population trends can be inferred from a single year's count, the USFWS, in their Southern Sea Otter Recovery Plan, recommend using 3-year running averages to assess overall population trends. This reduces the influence of potentially anomalous counts in a single given year.[97] Following these guidelines, the latest 3-year running average of the three most recent spring counts is up 8% over the previous average.[98] Although the 2005 census indicated a slight dip in the total otter counts compared to 2004, the results of the spring 2006 survey, which will begin in May, will shed further light on the status of the population and any potential changes in current trends.

c) **The otter population within Estero Bay is potentially near carrying capacity, and is therefore, not imbalanced.** The Revised Recovery Plan recognizes the importance of basing assessments of the southern otter population on "*…maximal levels relative to what the environment will support….*"[99] Similarly, the Marine Mammal Protection Act states that the goal for managing marine mammals should be to obtain an optimum sustainable population (OSP) "*keeping in mind the carrying capacity of the habitat.*"[100] Carrying capacity is the number of individuals that the resources of a habitat can support. Per the USFWS Recovery Plan, an OSP for the southern sea otter is "*likely a level equal to 50*

[97] USFWS Final Revised Recovery Plan for the Southern Sea Otter (Enhydra lutris) 2003.

[98] The Otter Project. USGS Spring 2005 News Release: *California Sea Otters -- 2005 Survey Numbers Dip But Overall Population Trend Remains Up (July 2005).*

[99] Final Recovery Plan, at 43.

[100] The Marine Mammal Protection Act of 1972 (as amended through 1997), Sec 2. Findings and Declaration of Policy 16 U.S.C. 1361, at 6.

to 80 percent of its current carrying capacity", or "approximately 8,400 animals for the entire California coast."[101] This figure is based on estimated historic population levels of 16,000 to 20,000 sea otters along the California coast.[102] At 2,735 individuals, the current total population of the southern otter[103] is at slightly less than 20% of the carrying capacity of 15,941 individuals.[104] Therefore, the southern otter as a whole has not reached an OSP along the central coast.

However, the distribution of otters, as well as habitat carrying capacities, is far from uniform along the central California coast. Otter densities and the associated carrying capacities of various habitats are dependent on a variety of factors, including differences in substrate and availability of prey refuges, with greater densities occurring in rocky-bottom habitats as compared to soft bottom habitats.[105] As such, the estimated carrying capacity for the sandy substrate region encompassing Estero and Morro Bay is significantly lower than that for the rocky areas and kelp beds to the immediate north and south.[106]

Using population data from 1996, maximum otter densities for sandy habitats are about 1.13 otters/km^2. This results in an estimated carrying capacity for the region of coastline between Cayucos and Hazard Canyon of approximately 89 otters.[107] However, census surveys have tallied more well over 100 independent individuals in this same region during both the spring and fall surveys for each of the last three years.[108] Even with an upward adjustment of maximum habitat densities to account for overall population increases since 1996, it is clear that the nearshore area from Cayucos to Morro Bay is near its carrying capacity for otters.[109] Furthermore, a population is thought to be at equilibrium with existing resources if the number of animals is not increasing and if environmental disturbances are not limiting further growth.[110] Thus while factors such as infectious disease play a role in sea otter mortality, they do not appear to be limiting the growth of the population in the Estero Bay region. Instead, the population is near equilibrium levels typical of the carrying capacity of sandy substrate areas.

d) **Higher stranding numbers in the Estero Bay area are expected, and they do not contradict the presence of an otter BIP.** USGS stranding numbers are highly dependent on factors unrelated to the actual mortality event, including 1) the potential for the carcass to be deposited within the study area, 2) the length of time the carcass persists on the shoreline once deposited, and 3) the likelihood that someone will see and report the stranding. Each of these factors serves to increase strandings reported in the region of Estero Bay compared to other locales.

The likelihood of carcass deposition is increased along the Estero Bay shoreline due to meteorological conditions and otter demographics. A study into patterns of carcass deposition conducted from 1980-1986 at Pt. Piedras Blancas indicates that "*several factors regulate the deposition of a carcass on*

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101 *Id.*, at E-7 (Response to Comment 21).
102 Laidre, K., R. Jameson, D. DeMaster, 2001. *An estimation of carrying capacity for sea otters along the California Coast.* Marine Mammal Science 17(2):294-309., at 305 (citing CDFG, 1976) California Department of Fish and Game. 1976. *A proposal for sea otter protection and research, and request for return of management to the state of California*. Unpublished 270 pp. Available from California Department of Fish and Game, Sacramento, Western Environmental Research Center, United States Geological Survey
103 The Otter Project, USGS spring 2005 otter survey summary USGS Spring 2005 News Release: *California Sea Otters -- 2005 Survey Numbers Dip But Overall Population Trend Remains Up* (July 2005)
104 The estimated statewide carrying capacity for the southern sea otter at 15,941 individuals, is consistent with the lower bound estimate of pre-exploitation abundance.
105 The Sea Otter (Riedman and Estes 1990).
106 Laidre et al 2001, at 300.
107 *Id*, at 301.
108 The Otter Project, USGS spring otter surveys, 2003-2005.
109 Laidre et al, 2001, at 307.
110 Estes, J.A. 1990. Growth and equilibrium in sea otter populations. Journal of Animal Ecology 59:385-401.

shore," including oceanographic and weather conditions, the abundance of live animals in proximity to the study area, and local physiography and beach orientation.[111] The long stretches of sandy, windswept, westward-facing beaches of Estero Bay are natural areas of deposition for floating debris carried southeastward by the prevailing winds. Otters that meet their demise along the high-population rocky shoreline to the northwest, within the southern portion of the San Simeon to Cayucos survey area, are likely to be deposited in the Estero Bay region due to the prevailing northwesterly winds. Once deposited, factors such as burial, scavenging, wave action, carcass size and stranding location may affect the persistence of a carcass.[111] Additionally, the longer a carcass persists, the more likely it is to be encountered by a human and subsequently reported. Otter carcasses washed up onto the wide sandy beaches surrounding Estero Bay are also likely to remain intact for longer periods than those that wash up on the adjacent rocky shorelines.

Because of these and other factors, the probability of reporting and recovery a stranded animal is inherently higher near Estero Bay than elsewhere along the coast. In its 2003 stranding summary, CDFG acknowledges that *"The probability of recovering stranded sea otters is greater in the Monterey Bay and Estero Bay regions than it is in most other areas in central California….*"[112] Three additional reasons for higher recovery rates along Estero Bay's beaches are the 1) increased visibility of carcasses, 2) accessibility of coastline, and 3) proximity to a substantial year-round human population. The otter's dark shading against the uniform background of light sand makes a stranding along Morro Strand State Beach or the Morro sandspit easily visible from a distance. Moreover, strandings along these beaches are more likely to be observed by beachgoers. Their ease of accessibility and proximity to urban population centers (Cayucos, Morro Bay, and Los Osos), as well as their status as regional points of interest (state beaches and state park lands), naturally results in higher visitation to these beaches than to most others along the central California coast. In contrast, few otter carcasses are recovered along the nearly 100 miles of coastline stretching from Cambria to Big Sur. Not only are there comparatively few people near the shoreline to potentially spot and report a stranded animal or carcass, but the rugged nature of the coastline limits both line-of-site for observing a stranding, and the ability to get close enough to positively identify a carcass.

Additionally, in the published stranding summaries reported by DCFG, the area of coastline between Cayucos and Hazard Canyon actually accounts for strandings along a larger area of coastline/habitat than other areas. The otter stranding network headed by CDFG assigns stranding locations based on 0.5-km increments along a smoothed California coastline (actually the 30 m isobath). However, this smoothed coastline does not account for the several miles of coastline within the 2,300 acres of open water within the Morro Bay Estuary at high tide.[113] Otters found stranded at any point within Morro Bay Estuary are automatically assigned to the 0.5 km position along the coast that coincides with the 30 m contour at the mouth of the Morro Bay. This results in an artifactual clustering of strandings at the mouth of the bay.

e) **The elevated stranding numbers noted by NRDC[114] for 2003 and 2004 do not demonstrate that a BIP is not present in the area surrounding the outfall.** Increases in stranding numbers in the region from Cayucos to Hazard Canyon are cited by NRDC as evidence that a BIP does not exist near the outfall. However, NRDC fails to recognize that the total strandings recorded for both 2003 and 2004

[111] Bodkin, J. L., and R. J. Jameson. 1991. Patterns of seabird and marine mammal carcass deposition along the central California coast, 1980-1986. Canadian Journal of Zoology 69:1149-1155. at 1153

[112] The Otter Project, USGS 2003 spring otter survey press release http://www.otterproject.org/site/pp.asp?c=8pIKIYMIG&b=1066903

[113] Haltiner, J. 1988. *Sediment Processes in Morro Bay, California*. Prepared by Philip Williams & Associates, Ltd. for the Coastal San Luis Resource Conservation District (CSLRCD) and the California State Coastal Conservancy (CSCC).

[114] NRDC, at 29.

(63 and 77 respectively) do not accurately reflect mortality rates within this same area.[115] Additionally, NRDC does not acknowledge that the population in this region is close to its carrying capacity; a population near its carrying capacity, is certainly not indicative of imminent extinction as claimed by NRDC.[116]

In addition, there are identifiable explanations for the increased strandings during 2003 and 2004 that are entirely unrelated to a BIP assessment or the MBCSD discharge. In his spring 2003 news release, USGS scientist Jim Estes indicates that the heightened mortality during 2003 may be related to the recent dramatic increase in the otter population as a whole.[117] He stated that "*The greatly elevated number of sea otters in Monterey Bay, and to a lesser extent in Estero Bay… may also help explain the record high number of strandings this year…*"[118] USGS personnel stated that "*Early storms and large waves during winter of 2002-2003 greatly reduced kelp canopies -- which otters use for resting and foraging -- in several exposed outer-coast areas…*" Additionally, USGS addressed the tendency for stranded otters to be preferentially recovered within the Estero Bay region.[112]

Insofar as 2004 stranding data, NRDC includes a table, which overdramatically highlights the 77 strandings recorded between Cayucos and Hazard Canyon over the course of the year. However, NRDC fails to mention that a single, highly unusual, stranding event in April 2004 accounted for almost half of these strandings.[119] During the month of April a total of 68 otters were found stranded statewide,[120] with 34 of these being in the area near Morro Bay. This number of strandings was unusual because it was over 3 times higher than the 10-year average. CDFG performed autopsies on 12 to 14 of the freshly deceased otters, and determined that almost all were infected with *Sarcocystis neurona*, a protozoan parasite, unrelated to *Toxoplasma gondii* .

S. neurona causes damage to the brain and other tissues in otters, and is responsible for equine protozoal myeloencephalitis in horses. It is commonly fatal in otters.[119] *S. neurona* is probably transmitted directly to otters through spoorocysts from opossum feces, because otters are not known to eat any of the known intermediates, such as birds. Because opossums are the only known reservoir (definitive host) for this parasite, it is thought that the 2004 stranding event was related to spring runoff events.[121] In the wake of the April 2004 event, sea otter mortality declined to more normal levels. Additionally, stranding data for 2005 did not demonstrate a repeat of the 2004 *S. neurona* event, and declined slightly from the 2004 high of 273 otters.[122]

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[115] The total stranding numbers for these two years represent almost half of the total population tallied for this same area during the spring surveys for both 2003 and 2004 (approximately 100 and 150 otters respectively). If the total numbers of strandings recorded for this section of coast all were caused by mortality within this same section of coast, one would arrive at the highly implausible conclusion that mortality for this area was around 50 percent. Rather, as discussed previously, many factors affect carcass deposition and eventual recovery; an otter carcass recovered from one section of coastline did not necessarily become ill or die there.

[116] NRDC, at 26. "*…it is well accepted that the otter population is likely to become "extinct" in the 'foreseeable future' in the vicinity of the outfall—Estero Bay—as well as throughout its limited 300-mile range*."

[117] The spring 2003 survey noted a significant increase in both independents and otter pups over 2002 counts. The Otter Project. USGS spring otter survey, 2003.

[118] USGS, WERC, 2003. News Release: California Sea Otter Numbers are Up for the 2003 Census. Accessed online, February 2006, at: http://www.werc.usgs.gov/news/2003-06-06.html

[119] CDFG. 2004. News Release: Scientists Determine April Sea Otter Deaths were associated with Brain Parasite, *Sarcocystis neurona*. Accessed online, February 2006, at: http://www.dfg.ca.gov/news/news04/04048.html

[120] Personal communication, Melissa Miller, February 20, 2006 concerning the details of the April 2004 *S. neurona* outbreak and stranding event.

[121] Personal communication, Melissa Miller, February 20, 2006

[122] Personal communication, Brian Hatfield, February 27, 2006 concerning 2005 otter stranding data.

f) **The home ranges of sea otters vary; otters residing in and near Estero Bay do not necessarily stay within the area year-round as stated by NRDC.**[123] Contrary to NRDC claims, more thorough readers of Riedman and Estes would notice that although sea otters may regularly stay within a small home range of approximately 1-2 km of coastline[124], they also make occasional long distance trips that take them well away from their normal haunts. This is because sea otter home ranges consist of several heavily used areas connected by travel corridors and that the ranges themselves vary in size depending on the location and quality of the range, as well as the sex and age of the otter. However, a general pattern observed in California for all age or sex classes was that individuals tended to remain in one area for extended periods with occasional sudden long-distance movements, which occurred year-round.[125]

Nevertheless, the fact that otters occasionally travel substantial distances makes them poor sentinels for evaluating impacts from point-source pollution. Specifically, it is impossible to determine where or when the sentinel marine organisms were impacted by exposure to pollution.[126] Because of this, sessile marine organisms, such as infauna and mussels, form the mainstay of monitoring programs around outfalls. Motile organisms, such as epifauna, fish, and mammals are simply not as diagnostic of potential impacts from a localized discharge.

Otter travel can be extensive and rapid. For example, adult male otters use two distinct home ranges connected by a migration corridor that can be traveled in a brief period of time as they traverse between female areas near the center of the range and male groups at the periphery of the range. These seasonal movements have been found to be as much as 60 to 100 km or more.[127]Additionally, males that live near the center of the range may journey farther than those already residing near the edges of the population. Like males, females are also capable of traveling long distances, and tagged females have been known to make round-trip journeys of up to 280 km.[127] However, compared to males, females have generally been found to make less extreme movements, and to travel intermediate distances. In California 7 of 13 adult females and 6 of 9 juvenile females moved maximum distances of >20 km.[127] Overall, however, though females generally have smaller lifetime ranges than males, they exhibit home ranges 1.5-2 times larger than resident males during the breeding season.[127]

g) **There are many causes of mortality in southern sea otters and *T. gondii* is not "…*the single most important known cause*", as erroneously implied by NRDC statements.** This is yet another case of NRDC inappropriately expanding on an otherwise legitimate quote from scientific literature. For example, NRDC begins by correctly citing the USFWS Final Revised Recover Plan for the Southern Sea Otter which states that "*Infectious disease is the single most important known cause of mortality among sea otters*."[128] However, NRDC then inappropriately embellishes this statement by attaching the specious claim, "*particularly encephalitis caused by the parasite Toxoplasma gondii (or "T. gondii")*." The Recovery Plan for the otter does not make that distinction. On the contrary, *T. gondii* is mentioned only once in the entire document, when it is listed together with a suite of other diseases.[129] The only infectious disease specifically mentioned in further detail within the Recovery Plan is acanthocephalan infection.[130]

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123 NRDC, at 7.
124 The Sea Otter, at 54-55
125 *Id*., at 54.
126 Miller et al 2004. *An unusual genotype of Toxoplasma gondii is common in California sea otters and is a cause of mortality*, International Journal of Parasitology 34(2004) 275-284; at 283.
127 *Id*., at 55
128 Final Revised Recovery Plan, at viii
129 Final Revised Recovery Plan, at 41 contains the following brief mention of *T. gondii*: "*Diseases, including acanthocephalan peritonitis, encephalitis, (caused by the protozoan Toxoplasma gondii, which completes its life cycle in cats and can occur in cat feces), coccidiomycosis, and various bacterial infections…*"
130 Final Revised Recovery Plan, at 7-9.

Without doubt, infectious diseases, including *T. gondii* infection, are currently contributing to mortality in sea otters. As cited by NRDC[131], *T. gondii* was found to be the primary cause of death in 16.2% of the 105 otter carcasses surveyed between 1998 and 2001 in a study on patterns of mortality in sea otters.[132] However, further reading in the same study, finds the following statement: "*Similar in proportionate mortality to T. gondii encephalitis,* ***infection with acanthocephalan parasites was a primary cause of death in 16.2% of otters examined****…*" (emphasis added). The study determined that "*encephalitis due to T. gondii was one of the two leading causes of mortality identified in otters during the time period studied*" (1998-2001), and clearly identifies *T. gondii* and acanthocephalan infection as being similar in their contributions to mortality, including sharing the same percentage of primary mortality attributable to each cause.[133] However, the misplaced emphasis on *T. gondii* in the statement by NRDC, and the omission of the conclusion that both causes contributed to mortality in approximately the same amounts, seeks to deliberately mislead the reader. Here, NRDC selectively spliced quotations together to imply a conclusion that it is clearly not supported by the scientific data.

h) **The presence of a BIP within the Morro Bay Estuary is not germane to an evaluation of the MBCSD discharge within Estero Bay.** Contrary to the NRDC statements, an assessment of a BIP within the Morro Bay Estuary in addition to Estero Bay is not germane to an evaluation of the discharge.[134] They claim that "*…it is undisputed that the…Plant discharges an average of 1.4 million gallons of freshwater wastewater into Morro Bay every day….*"[17] This is ridiculous. The discharge is into Estero Bay, not the Morro Bay Estuary. Furthermore, as described in Response 2 on Page 6 of this letter, the plume-tracking study cited by NRDC to support their claims[135] did not demonstrate that MBCSD effluent actually enters the Morro Bay Estuary. Even if the findings of this seriously flawed tracking study were correct, wastewater incursions into the Estuary would be extremely rare and so phenomenally dilute (50,000:1) that there could be no conceivable impact on estuarine biota. Nevertheless, any potential *T. gondii* input into the estuary from the MBCSD discharge would be completely negligible compared to the freshwater pathogen sources that discharge directly into the estuary.[23]

i) **The presence of a threatened species does not preclude the existence of a BIP.** NRDC[136] incorrectly asserts "*…the otters' threatened listing functions as per se evidence that a balanced indigenous population of marine life is not present.*" They base their assertion on an incorrect definition of "*threatened species*" under the Endangered Species Act (ESA). The ESA defines an "*endangered"* species as one that is "*in danger of extinction throughout all or a significant portion of its range*", while a *threatened* species is one that is likely to become endangered (due to a small base population, habitat destruction, etc.) within the foreseeable future, but is not currently facing extirpation[137]. NRDC's statement[138] that the otter was listed "*because it is likely to become endangered (**i.e. extinct**)*" (emphasis added) is highly inaccurate as it suggests that endangered is synonymous with extinct.

The otter was originally listed as a "threatened" species under the ESA because of its "*small population size, its limited distribution, and potential risk to its habitat and population from oil spills.*"[139] At the time of its listing, concerns existed that a single oil spill could wipe out much of the population given

131 NRDC, at 9, 29.
132 Kreuder D. et al. 2003. *Patterns of Mortality in southern sea otters (Enhydra lutris nereis) from 1998-2001* Journal of Wildlife Diseases 495 (2003), at 499.
133 *Id.* at 499.
134 NRDC, at 20: "*…staff have failed to fully analyze the issue of whether a "balanced indigenous population" of marine life exists in Estero Bay* ***and Morro Bay****,*" emphasis added.
135 Anthony et al. 1986. Morro Bay Bacterial Study 1986-1987.
136 NRDC, at 26.
137 Endangered Species Act of 1973, Section 3.
138 NRDC, at 26
139 http://www.epa.gov/fedrgstr/EPA-IMPACT/2004/February/Day-06/i2558.htm

its small size and localized distribution. Since 1977, however, the population has more than doubled in size, and the distribution of the otter population has expanded to both the north and south. In summary, although the otter population still has a long way to go in order to attain OSP and historical population levels, it is not currently facing extinction as erroneously implied by NRDC. Therefore, the *threatened* listing of the otter is not "*dispositive*" of the existence of a BIP in the region surrounding the outfall as asserted by NRDC, nor does its *threatened* status "*function as per se evidence*" that a BIP is not present in the area.

Please contact the undersigned if you have questions regarding these responses.

Sincerely,

Douglas A. Coats
Vice President
Marine Research Specialists
Dr. Douglas A. Coats
2006.03.03 16:18:33 -08'00'

Douglas A. Coats, Ph.D.
Program Manager

Attachments: Curriculum Vitae - Coats
2005 MBCSD Annual Report

EDUCATION

B.S.	Physics, California State Polytechnic University	1975
M.S.	Oceanography, Scripps Institution of Oceanography	1979
Ph.D.	Oceanography, Scripps Institution of Oceanography	1982

SUMMARY OF EXPERIENCE

Dr. Coats is a marine scientist with over 30 years of experience. After acquiring formal academic training in all aspects of chemical, physical, geological, and biological oceanography at Scripps Institution of Oceanography, he focused on resolving complex environmental issues related to coastal development. He is known for establishing industry-wide (API) design procedures as a result of wave-propagation studies that he directed at the California Institute of Technology. He authored over 20 reports specifying site-specific environmental design criteria offshore California, Japan, Alaska, Norway, and Australia. In addition, he has acted as an expert witness at California Coastal Commission Hearings and as a scientific advisor and member of the steering committee for the Royal Norwegian Council for Scientific and Industrial Research in Oslo, Norway.

As Senior Oceanographer at Marine Research Specialists, Dr. Coats has been the Principal Investigator responsible for measurement and interpretation of coastal marine processes in a number of large, high profile, multi-disciplinary programs. For example, the MMS-sponsored California Monitoring Program was conducted to assess potential impacts of Oil and Gas activities offshore Pt. Conception. By assimilating measurements from nearly every aspect of the program in a resuspension and trajectory model, Dr. Coats accurately determined suspended sediment loads as well as their drilling-derived components. Results were verified with the field data, which included measurements of daily drilling-mud emissions from platforms, sediment trap volumes, surficial sediment chemistry, and current velocities.

Shortly thereafter, Dr. Coats was the Principal Investigator responsible for the analysis of all benthic data collected in a large multi-year field program to monitor municipal discharges in Massachusetts Bay for the Water Resources Authority. Analyses of baseline physical, chemical and biological data in surficial sediments led to his formulation of testable hypotheses concerning potential impacts from large particulate loads discharged from a municipal outfall in the Bay. His analyses included modeling of the projected contaminant increases in surficial sediments due to the transport of effluent particulates. From the projected contaminant loading, he estimated the localized biological impacts by applying recently-developed biostatistical analyses. He also performed a quantitative investigation of hard-substrate features from photoimages collected by ROV along seven miles of tracklines. Other discharge-related experience included an environmental assessment of waste discharge into the coastal waters in the southern California Bight. In addition, Dr. Coats generated technical memoranda on regulatory compliance of the Los Angeles Clean Water Program Master Plan to NEPA, CEQA and the Clean Water Act. These included assessments of regional water quality, marine biology, and endangered species. He also prepared environmental documents characterizing projected water quality at several candidate outfall sites in conjunction with the San Diego Metropolitan Clean Water Project and assessed NPDES monitoring data collected in San Francisco Bay as part of the East Bay Dischargers Authority.

More recently, Dr. Coats has prepared numerous marine biology and marine water quality sections for major environmental impact assessments, reports, and studies. High profile projects on which Dr. Coats has previously worked on include the Carpinteria Field Redevelopment (California State Lands Commission), Tranquillon Ridge Oil Development (Santa Barbara County), Guadalupe Oil Field Remediation (San Luis Obispo County), Abandonment and Unocal Avila Beach Cleanup (San Luis Obispo County), A Survey of Prominent Anchor Scars and the Level of Disturbance to Hard-Substrate Communities in the Point Arguello Region (Chevron). Most of the marine assessments dealt with controversial projects such as oil spill remediation activities, coastal development, point-source discharges, offshore fiber-optic cable installation, and the development of offshore oil fields. All of the projects required definitive analyses of available biological and water-quality data in order to support conclusions as to the significance of potential impacts. In a number of cases, Dr. Coats also conducted offshore surveys to fill in existing data gaps. He was responsible for developing quantitative significance criteria and mitigation measures based on regulatory limits that have became the standard in subsequent EIRs. The significance criteria he developed for coastal remediation and construction projects has been adopted verbatim by county, state, and federal agencies for other projects under their purview.

In San Luis Obispo County, Dr. Coats was the Principal Investigator for marine water quality in the environmental evaluations conducted as part of oil spill remediation activities at Avila Beach and Guadalupe, California. This work was conducted on behalf of local agencies and the Regional Water Quality Control Board. He examined projected water quality impacts and their influence on marine fauna of the region and was responsible for preparing the oceanographic sections on these two complex and controversial EIRs. In addition, he contributed to EIR sections dealing with marine biological resources and prepared detailed analyses of dispersion and chronic effects of long-term marine contamination. As part of the environmental assessment, Dr. Coats conducted field surveys of the intertidal and estuarine environments at both locations. Finally, because both sites are adjacent to wetland regions, he was responsible for evaluating models of tidal inlets as they pertain to wetland and estuary management.

Dr. Coats has additional experience in the assessment of marine impacts along the central California coast where he was the Principal Investigator for marine resources in the environmental evaluation conducted as part of the installation of fiber-optic cables offshore San Luis Obispo County. For this EIR, he conducted a number of offshore surveys to augment historical data. He also specified quantitative significance criteria to assess impacts and developed mitigation measures that became the standard for all subsequent EIRs related to numerous other cable installations proposed for this section of the central California coast. Dr. Coats additionally developed offshore monitoring techniques to identify and limit the release of drilling mud and other contaminants into the marine environment during directional drilling. These techniques are now required during directional drilling of all fiber-optic cable conduits offshore California.

Dr. Coats is currently the Program Manager of an ongoing study on the recovery of intertidal environment in the wake of the Exxon Valdez oil spill in Prince William Sound, Alaska. This ongoing study, conducted under the auspices of NOAA, is now entering its 10^{th} year. Dr. Coats has been responsible for the application of innovative statistical techniques to the long-term environmental data acquired in this study in order to quantitatively measure recovery of all aspects of the biological and physical environment. In addition to his authorship of a number of NOAA reports, over the years his collaborative research efforts have culminated in the publication of several articles in the peer-reviewed scientific literature.

Finally, Dr. Coats has been the marine environmental consultant to the City of Morro Bay since 1993. He is primarily responsible for National Pollutant Discharge Elimination System (NPDES) monitoring associated with the City’s 301(h)-modified effluent discharge into the open coastal waters of Estero Bay. In this capacity, Dr. Coats has collected, analyzed, and interpreted a plethora of oceanographic data during the regular chemical, biological, and physical surveys conducted on the receiving waters, benthic sediments, effluent, sludge, and treatment plant processes. As a result he has assimilated more than two decades of monitoring data in investigations and analyses of potential impacts from the discharge.

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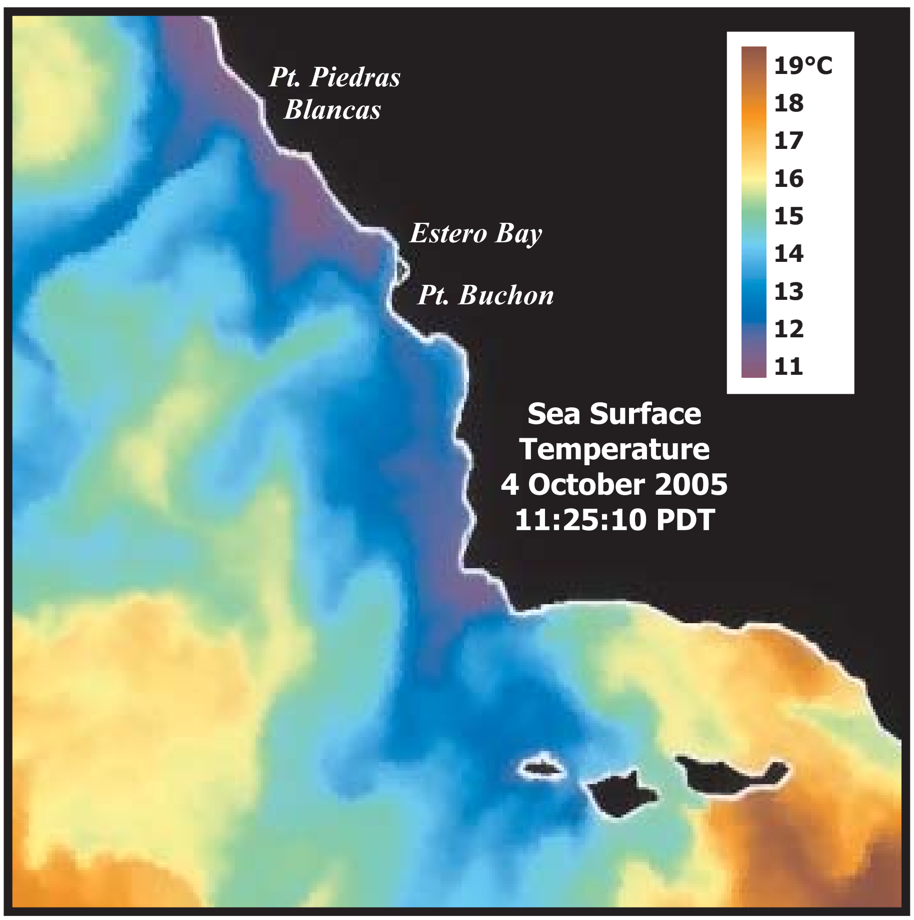
City of Morro Bay and
Cayucos Sanitary District

OFFSHORE MONITORING AND REPORTING PROGRAM

2005 ANNUAL REPORT



Marine Research Specialists

3140 Telegraph Rd., Suite A
Ventura, California 93003

Report to

City of Morro Bay and
Cayucos Sanitary District

955 Shasta Avenue
Morro Bay, California 93442
(805) 772-6272

OFFSHORE MONITORING AND REPORTING PROGRAM

2005 ANNUAL REPORT

Prepared by

Douglas A. Coats and Bonnie Luke
(*Marine Research Specialists*)

and

Bruce Keogh
(*Morro Bay/Cayucos Wastewater Treatment Plant*)

Submitted by

Marine Research Specialists
3140 Telegraph Rd., Suite A
Ventura, California 93003

Telephone: (805) 644-1180
Telefax: (805) 289-3935
E-mail: Doug.Coats@mrsenv.com

February 2006

marine research specialists

3140 Telegraph Rd., Suite A · Ventura, CA 93003 · (805) 644-1180

Mr. Bruce Keogh
Wastewater Division Manager
City of Morro Bay
955 Shasta Avenue
Morro Bay, CA 93442

15 February 2006

Reference: 2005 Annual Monitoring Report

Dear Mr. Keogh:

Enclosed is the referenced report. It documents the continued effectiveness of the treatment process, the absence of marine impacts, and compliance with the discharge limitations and reporting requirements specified in the NPDES discharge permit.

Please contact the undersigned if you have any questions regarding this report.

Sincerely,

Douglas A. Coats, Ph.D.
Project Manager

Enclosure (Seven Copies)

I certify under penalty of law that this document and all attachments were prepared under my direction or supervision in accordance with a system designed to assure that qualified personnel properly gather and evaluate the information submitted. Based on my inquiry of the person or persons who manage the system, or those persons directly responsible for gathering the information, the information submitted is to the best of my knowledge and belief, true, accurate and complete. I am aware that there are significant penalties for submitting false information, including the possibility of fine and imprisonment for knowing violations.

Mr. Bruce Ambo
City of Morro Bay

Date

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LIST OF ACRONYMS AND DEFINITIONS

ACOE	**U.S. Army Corps of Engineers**
Anomaly	**A deviation or departure from ambient (mean) conditions in one of the six measured seawater properties: salinity, temperature, density, dissolved oxygen, alkalinity, or transmissivity. The amplitude of the anomaly is quantified as the difference between the measured seawater property at given station and the mean of that property measured at all other stations at the same depth level or distance above the bottom.**
Anthro-pogenic	**Changes to the environment induced by human activities**
BCCE	**Brown and Caldwell Consulting Engineers**
BEHP	**Bis 2-Ethyl Hexyl Phthalate is a ubiquitous compound in the environment that leaches out of plastics.**
BNL	**Bottom nepheloid layer is a common phenomenon on continental shelves where increased turbidity near the seafloor is caused by the presence of naturally occurring particulates formed from light-weight flocs of detritus. These flocs are easily suspended by oscillatory bottom currents generated by passing surface gravity waves.**
BOD_5	**Five-day biochemical oxygen demand**
BTEX	**Benzene, toluene, ethyl benzene, and xylene**
Cal/OSHA	**California Department of Industrial Relations, Division of Occupational Safety and Health**
CAM-17	**CAM is an acronym for California Administrative Manual and CAM-17 refers to the list of heavy metals identified in the California Code of Regulations (CCR 2003), Title 22, Chapter 11: Identification and Listing of Hazardous Waste.**
CEQA	**California Environmental Quality Act**
CCR	**California Code of Regulations (CCR 2003)**
CDFG	**California Department of Fish and Game**
COP	**The California Ocean Plan (SWRCB 1997) has been revised but the 1997 edition was in force when the current NPDES discharge permit was issued to MBCSD.**
CTD	**An instrument package that continuously records Conductivity, Temperature, and Depth, in addition to a number of other physical oceanographic parameters in the water column.**
CV	**A coefficient of variation is used to compare the relative amounts of variation in populations having different means. It is defined as the standard deviation expressed as a percentage of the mean.**
DDT	**Dichloro Diphenyl Trichloroethane, a chlorinated pesticide.**
DHS	**California Department of Health Services**
DMR-QA	**Discharge Monitoring Report Quality Assurance (DMR-QA) Study administered by the United States Environmental Protection Agency (USEPA)**

LIST OF ACRONYMS AND DEFINITIONS
(Continued)

DO	**Dissolved Oxygen**
ELAP	**California Department of Health Services, Environmental Laboratory Accreditation Program**
EQ	**Exceptional Quality (EQ) compost that meets the Federal and State requirements for beneficial reuse in the local community.**
ERL	**Effects Range Low: a sediment toxicity concentration below which adverse biological effects are not expected to occur**
ERM	**Effects Range Median: a sediment toxicity concentration above which adverse biological effects are expected to occur**
GPM	**Gallons per Minute**
Hazwopper	**Hazardous Waste Operations and Emergency Response**
HCl	**Hydrochloric Acid**
HCH	**Isomers of hexachlorocyclohexane including alpha, beta, gamma (lindane), and delta**
I&I	**Inflow and Infiltration**
Isopycnal	**A surface of constant seawater density**
Leachate	**A solution formed by leaching, especially a solution containing contaminants picked up through the leaching of soil.**
LC_{50}	**Lethal 50% Concentration: A toxicity endpoint where 50% of the specimens will die after a 96-hr exposure period**
MBCSD	**The City of Morro Bay-Cayucos Sanitary District**
MTBE	**Methyl Tertiary Butyl Ether, an additive to gasoline that boosts octane to reduce air emissions but has contaminated groundwater in many regions after accidentally leaking from underground storage tanks.**
MDL	**Method Detection Limit, which is the lowest concentration that can be reported under ideal conditions where the sample contains only the compound of interest with a concentration in an optimal calibration range and in a medium that does not interfere with the performance of the analytical instrument.**
MGD	**Million Gallons per Day**
mg/Kg	**Milligrams per Kilogram = μg/g dry weight = parts per million (ppm)**
mg/L	**Milligrams per Liter = aqueous parts per million (ppm)**
Mixed liquor	**A mixture of activated sludge, wastewater, and oxygen where organic sewage constituents are biologically assimilated**
MLLW	**Mean Lower Low Water**
MMS	**Minerals Management Service**
MPN	**Most Probable Number**

LIST OF ACRONYMS AND DEFINITIONS
(Continued)

MRS	**Marine Research Specialists**
MT	**Metric Ton = 1,000 kg**
NEP	**National Estuary Program**
NOAA	**National Oceanic and Atmospheric Administration**
NOEC	**No Observable Effect Concentration**
NPDES	**National Pollutant Discharge Elimination System**
NTU	**Nephelometric Turbidity Units**
O&G	**Oil and Grease**
OIT	**Operator-in-Training**
P1 – P12	**Individual identifiers for discharge-related perturbations in seawater properties**
PAH	**Polynuclear Aromatic Hydrocarbons**
PCA	**Principal Component Analysis is a multivariate technique that arranges samples along an axis based on species composition. This arrangement is conducted independently along a number of dimensions (usually 2 or 3) that approximate some pattern of response of the intertidal community to underlying environmental gradients. PCA condenses the complex species-abundance database to a few factors responsible for observed variability within the intertidal community structure, while retaining ecologically meaningful biological information.**
PCB	**Polychlorinated Biphenyl**
PCi/L	**Pico-Curies per liter, a measure of aqueous radioactivity**
Perturbation	**A small but detectable variation (anomaly) in one or more seawater properties caused by the presence of dilute effluent within the receiving waters. Individual Perturbations identified in this report are designated P1 through P12.**
PG&E	**Pacific Gas and Electric**
POTW	**Publicly-Owned Treatment Works**
Power (1-β)	**The probability correctly rejecting the null hypothesis depending on the significance criterion, the variability of the sample results, and the size of the impact.**
ppm	**Parts per million = mg/L in solution , or mg/Kg = μg/g dry weight**
PQL	**Practical Quantification Limit, which is the lowest concentration that can be measured with statistical reliability given the sample size and analytical method.**
PSDWF	**Peak seasonal dry-weather flow**
***p*-value**	**Low *p*-values (<0.05) indicate a high degree of confidence that the computed quantity is statistically significant.**
PWWF	**Peak wet-weather flow**
Pycnocline	**A region of rapid vertical change in the density field of the seawater column**

LIST OF ACRONYMS AND DEFINITIONS
(Continued)

QA/QC	**Quality Assurance and Quality Control**
RAS	**Return Activated Sludge is returned continuously to the aeration tank to provide a seed of developed bacteria for rapid digestion of incoming wastewater. Some sludge is wasted (WAS) to prevent excessive solids buildup.**
RWQCB	**State of California Regional Water Quality Control Board - Central Coast Region**
SAIC	**Science Applications International Corporation**
SBE	**Sea-Bird Electronics, Inc.**
SCBPPFCT	**Southern California Bight Pilot Project Field Coordination Team**
SCCWRP	**Southern California Coastal Water Research Project**
SEP	**Supplemental Environmental Project**
Sigma-T	**σ_t units are a measure of the density of water equal to the 1000 times the specific gravity minus one**
SJCF	**San Joaquin Composting Facility located in Kern County California and owned by McCarthy Family Farms, Inc.**
STLC	**Soluble Threshold Limit Concentration applies to the measured concentration in the liquid extract from a biosolid sample, as determined by a Waste Extraction Test or WET. Biosolids with leachate concentrations exceeding the STLC are classified as hazardous in the State of California.**
SWRCB	**State Water Resources Control Board**
Thermocline	**A region of rapid vertical change in the temperature field of the seawater column**
TIC	**Tentatively Identified Compound TIC is a compound that can be seen by the analytical testing method, but its identity and concentration cannot be confirmed without further analytical investigation. Many analytical methods can report TICs but the analysis is not targeted specifically for their detection.**
TKN	**Total Kjeldahl Nitrogen**
TRC	**Total Residual Chlorine in effluent determined from grab samples collected beyond the chlorine contact chamber**
TSS	**Total Suspended Solids**
TTLC	**Total Threshold Limit Concentration that applies to the total wet-weight concentration of a contaminant within a bulk biosolid sample consisting of the entire millable solid matrix rather than just the leachate. Biosolids are designated as hazardous wastes in the State of California if measured bulk concentrations exceed the TTLC.**
TUa	**Acute Toxicity Units**
TUc	**Chronic Toxicity Units**
TVS	**Total Volatile Solids**

LIST OF ACRONYMS AND DEFINITIONS
(Continued)

USEPA	**U.S. Environmental Protection Agency**
USGPO	**U.S. Government Printing Office**
USMMS	**U.S. Minerals Management Service**
VFD	**Variable-Frequency Drives are electronic motor-speed controllers used at the MBCSD WWTP to precisely control the influent pumping rate at the plant headworks by adjusting the frequency of electrical power sent to the pump motors.**
WAS	**Waste Activated Sludge is removed from the treatment process to prevent excessive solids buildup.**
WET	**Waste Extraction Tests measure the soluble leachate or the extractable amount of a substance contained within a bulk sample of biosolids. A WET is indicated if the bulk wet-weight concentration of a contaminant in a biosolids sample exceeds ten times the STLC.**
WWTP	**City of Morro Bay-Cayucos Sanitary District Waste Water Treatment Plant**
Zeolite	**An inorganic porous material having a highly regular structure of pores and chambers that allows some molecules to pass through, and causes others to be either excluded, or broken down**
ZID	**Zone of Initial Dilution is a limited volume of water surrounding the outfall where wastewater rapidly mixes with receiving waters. Most receiving-water objectives of the Ocean Plan do not apply within the ZID.**

EXECUTIVE SUMMARY

The City of Morro Bay and the Cayucos Sanitary District (MBCSD) jointly own the wastewater treatment plant operated by the City of Morro Bay. A National Pollutant Discharge Elimination System (NPDES) permit, modifying secondary treatment requirements, was issued in December 1998. Under the authority granted by the permit, the treatment plant discharges effluent to the open ocean. The permit allows discharge of blended primary and secondary-treated wastewater, although vast majority of the wastewater receives secondary treatment. The discharge permit requires a monitoring and reporting program to evaluate short- and long-term effects of the effluent discharge on receiving waters, benthic sediments, and infaunal communities. This 2005 Annual Report is submitted in partial satisfaction of those reporting requirements.

This document presents a comprehensive analysis of the extensive monitoring data collected over the last two decades. Virtually every aspect of the treatment process, receiving waters, and seafloor sediments was monitored. An exhaustive quantitative analysis of all measured parameters demonstrates that the effluent discharge consistently meets the discharge requirements, and has no discernible effect on the ocean environment. A comparison of the properties of the influent and effluent affirms the treatment plant's proficiency in removing contaminants and reducing organic loads within the wastewater stream. All offshore receiving-water measurements showed that the effluent plume was largely restricted to a narrow 15-m zone-of-initial-dilution around the outfall. Measurements within the effluent plume within a few feet of a diffuser port quantified the plume's rapid dispersion, and demonstrated that the structure was operating better than predicted by modeling. Finally, the absence of adverse discharge-related impacts to the physics, chemistry, and biology of benthic sediments verified the effectiveness of the treatment process, the high dilution of effluent within receiving waters, and the low toxicity of the discharged effluent. These conclusions were supported by an independent review of monitoring data collected prior to 1998 that was conducted by the United States Environmental Protection Agency (USEPA 1998). The additional data presented in this report are consistent with the historical data insofar as the treatment plant's continued low mass-emission of contaminants, low toxicity of the effluent stream, and absence of impacts to the marine environment.

The plant operates under a 301(h)-permit that modifies only the NPDES requirements on suspended-solid and biochemical oxygen demand (BOD) emissions. All other NPDES limits, including restrictions on the discharge of toxic substances, apply to the MBCSD discharge without exception. However, the partial-secondary treatment currently performed by the plant routinely achieves reductions in suspended solids and BOD that are close to, and often exceed, full secondary requirements. In fact, the effluent usually consists entirely of wastewater that has received secondary treatment. Blending of primary- and secondary-treated wastewater is usually only required during periods of particularly high flow, such as during rainstorms. During 2005, the blending valve was open only 7.5% of the time.

Effluent monitoring during 2005 documented a year of exceptionally high operational performance by the treatment plant. Major effluent constituents, including suspended solids, BOD, and oil and grease (O&G) all had much lower concentrations and mass emissions than the permitted maximums, as has been the case in prior years. However, in 2005, the annual averages of several key diagnostics of treatment performance were some of the best reported in nearly two decades of monitoring. The suspended solids concentration within effluent was the second lowest on record, while the solids removal rate was the highest on record. Combined with the low total flow, the plant's solids-removal process resulted in the lowest annual mass emission of suspended solids recorded since monitoring began in 1986. Similarly, the aver-

age effluent O&G and BOD concentrations in 2005 were much lower than average despite higher-than-average influent concentrations.

Some of the reduction in mass emissions was attributable to the lower overall flow in 2005 compared to prior years. In the past few years, flow reductions were achieved by successfully eliminating major sources of rainwater inflow and groundwater infiltration into the collection system. In addition, a more accurate flow meter was commissioned in 2002. As a result of these efforts, the annual average flow reported in 2005 was one of the lowest on record. Despite the low flow in 2005, the plant continued to remove major wastewater constituents from the influent at consistently high rates.

Although the treatment process efficiently removed major organic wastewater constituents such as suspended solids, BOD, O&G, and coliform bacteria, the general lack of other chemical contaminates within effluent was largely due to their absence within the influent stream. The treatment plant is not designed to extract heavy metals and synthetic organic compounds, such as pesticides and PCBs, from wastewater. Instead, the lack of heavy industry within the service area is largely responsible for the effluent's low toxicity. The few businesses that discharge to the sewer system produce wastewater similar to that from domestic sources, only on a larger scale. These sources are documented in a comprehensive industrial waste survey that is maintained and periodically updated by plant personnel. In addition, an ongoing public-outreach program and the convenience of an onsite household hazardous waste recycling facility further reduced the introduction of pollutants into the waste stream. Since its opening in August 2000, the household hazardous waste disposal site at the Wastewater Treatment Plant has become one of the most heavily used hazardous-waste receiving facilities in San Luis Obispo County. During 2005. the permanent hazardous waste facility collected over 73 tons of paint, flammable liquids, antifreeze, car batteries, motor oil, and hazardous materials that otherwise may have been discharged to the ocean or ended up in a landfill. The collection facility also now accepts electronic waste such as television and computer monitors. This new effort helps to address long-standing problems with the improper disposal of these components.

Over the past two decades of operation, the treatment plant has consistently outperformed expectations for wastewater treatment based on regulatory standards. During this time, there has been no indication of deterioration in plant performance, and effluent quality has consistently exceeded the performance criteria anticipated in the original design. On rare occasions where exceptions to standards or criteria have occurred, they have been the direct result of unforeseen external events, or brief, unavoidable mechanical failure in a treatment-system component. In spite of a diligent program of preventative maintenance, the challenge for plant personnel has been to quickly respond to unanticipated failures in system components.

Among the thousands of samples and measurements collected as part of the monitoring program during 2005, there were no exceptions to the waste-discharge requirements specified in the NPDES permit. This perfect level of compliance was unusual for the MBCSD treatment plant, given that unavoidable external events and unforeseen mechanical malfunctions have occasionally affected the treatment process in past years. The compliance record achieved in 2005 reflects the culmination of proactive measures that have been regularly implemented by knowledgeable and experienced MBCSD personnel over many years of operating this facility.

A new more-rigorous offshore-monitoring program was instituted when the NPDES permit was re-issued in 1998. It effectively doubled the number of water-quality samples collected close to the outfall, and significantly improved the ability to detect potential discharge-related impacts in the marine environment. All of the receiving-water stations were relocated to within 100 m of the diffuser structure. Additionally, four new benthic sediment stations were positioned 60 m from the diffuser, providing a six-fold increase in the ability to detect contaminant accumulations. Despite the increased spatial resolution, no evidence of adverse discharge-related marine impacts was found during 2005. As has been the case throughout the two decades of monitoring, all measurements fully complied with requirements of the discharge permit and the objectives of the California Ocean Plan.

Receiving-water quality was evaluated with quarterly offshore water-column surveys. The high-precision measurements collected during the surveys were capable of resolving minute changes in ambient water properties. They were used to detect and delineate the effluent as it mixed with receiving waters upon discharge from the diffuser. Slight anomalies in water properties associated with the presence of dilute effluent were observed near the zone of initial dilution (ZID). These highly localized anomalies were within the limits specified in the NPDES permit. Moreover, the high degree of effluent dilution that was observed in the anomalies close to a diffuser port attests to the effective and efficient operation of the diffuser structure in rapidly mixing the wastewater within receiving waters. The limited spatial extent of measurable effluent perturbations should also resolve any concerns over potential impacts to the waters of Morro Bay, which is designated as a national marine estuary. Even with the use of highly sensitive instrumentation, the presence of effluent constituents could not be detected more than 61 m beyond of the diffuser structure. Consequently, it is inconceivable that the discharge could materially affect the sheltered Bay waters lying 2.8 km to the south.

Sampling of the surfzone for coliform bacteria was another component of receiving-water monitoring during 2005. Surfzone samples were collected at seven locations along Atascadero State Beach. Few of the samples contained measurable quantities of coliform organisms, and elevated densities were associated with non-point source contamination rather than the discharge of effluent. For example, consistently high densities (>1600 MPN/100-ml) of total and fecal coliform organisms were detected at the mouth of Morro Creek during periods when it was flowing into the ocean. Other areas of the beach seldom showed elevated coliform levels. The few instances of elevated coliform at other stations tended to be isolated measurements rather than repeated instances of increased coliform at a given station. Consequently, the surfzone measurements met the stringent shellfish harvesting limitation, which is based on a monthly median. The general absence of detectable coliform levels within discharged effluent during 2005 confirmed the effectiveness of the treatment plant's disinfection process at removing these microbial contaminants from the waste stream. None of these instances of elevated coliform coincided with the rare occasions when coliform was also slightly elevated in discharged effluent. Instead, the elevated beach coliform measurements either: 1) were measured at distant reference stations, 2) coincided with rainfall events, 3) were proximal to shoreline organic debris sources, or 4) consisted mostly of non-fecal coliform.

The physics, chemistry, and biology of benthic sediments around the outfall have also been monitored for two decades. Benthic environments are important indicators of the presence of marine pollution because they act as a major reservoir for most contaminants that enter the ocean. The lack of perceptible impacts to the benthic environment around the diffuser structure during 2005 confirmed that the treatment process effectively removed contaminants from the influent stream and that the diffuser structure efficiently diluted wastewater upon its discharge into receiving waters.

Three sediment-chemistry analyses documented the absence of adverse discharge-related benthic impacts. First, chemical concentrations measured within Estero Bay sediments during 2005, and in prior years of monitoring, were below thresholds that have been identified as toxic to marine organisms. This includes samples collected close to the diffuser structure, as well as those found more than 1 km away. In fact, measured concentrations were comparable to those found in benthic sediments collected throughout the region, but were generally much less than those found within the Southern California Bight. Second, benwhich is an artifact of the new bioassay protocols that were instituted in April 2003. Nevertheless, all toxicity endpoints were below permitted limits.

Finally, the effectiveness of the treatment plant was demonstrated by its ability to accommodate unusual flow events and unforeseen mechanical upsets. Eleven anomalous plant conditions were identified in the monitoring data during 2005. None of these events were related to random perturbations in the efficiency of wastewater treatment. On the contrary, all were directly attributable to external events or necessary repairs to treatment-plant equipment. Six events were perceptible increases in plant inflow during rainstorms. Four of these six rainfall events coincided with slight but perceptible increases in plant throughput due to population increases during holidays. Four additional increased-flow events were solely associated with increased tourist populations during a holiday. The remaining event occurred when improvements to the disinfection system piping required manual infusion of chlorine, which resulted in chlorine concentra-

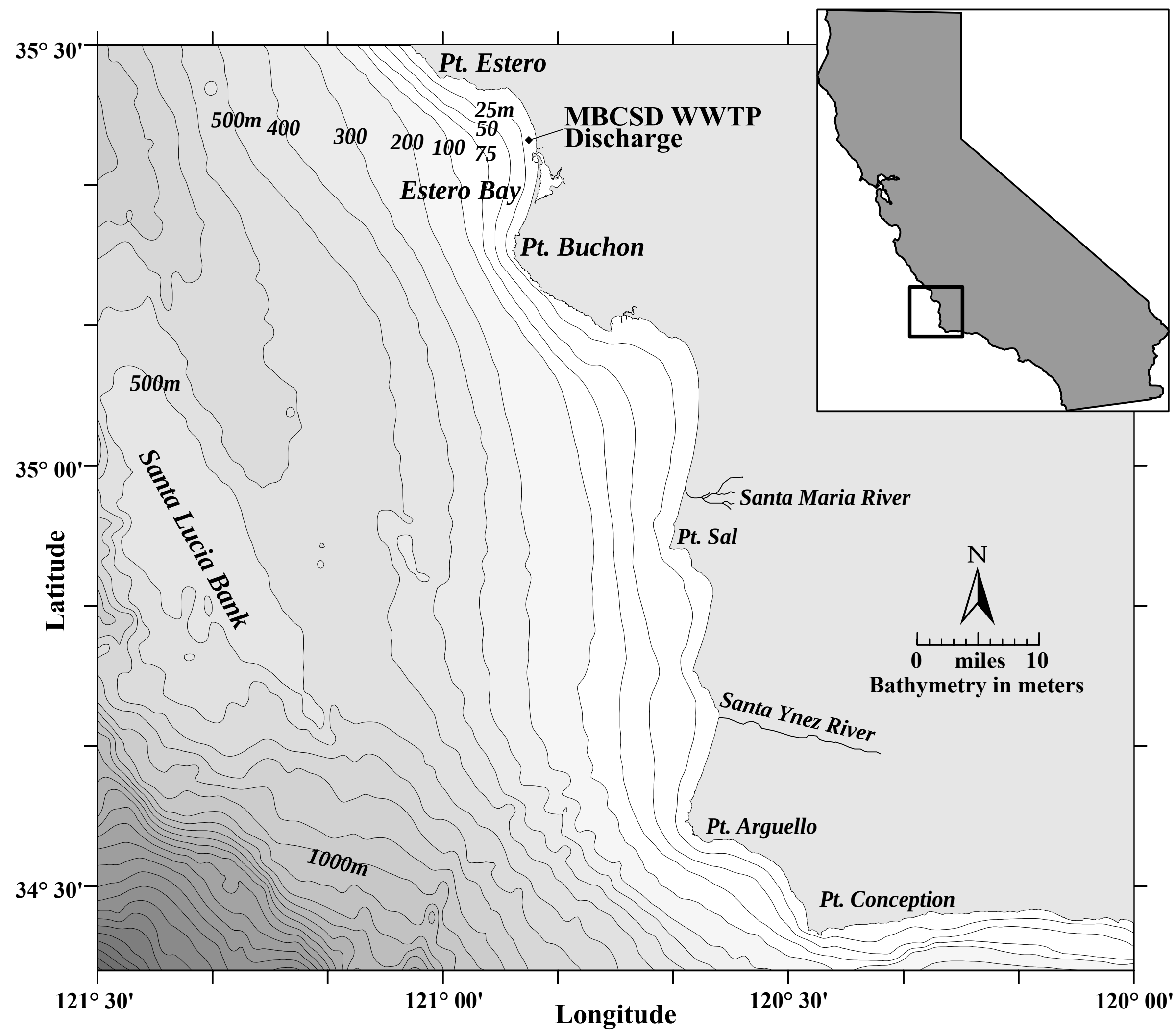

Figure 2.1 Regional Location of the Treatment Plant

thic samples collected during 2005 did not exhibit any statistically significant gradient of consistently increasing contaminant concentration with increasing proximity to the outfall. Finally, although the concentrations of some metals have exhibited a slight but steady increase over time, the increase is unrelated to effluent discharge. The volume of metals discharged within effluent was too small to account for the slight increase in sediment concentrations that has been observed throughout Estero Bay. Moreover the increase is unrelated to outfall proximity. Finally, the trace-metal concentrations measured in the bulk sediment samples arise from metal that is bound into the mineral matrix of sand grains, where it has little affect on marine organisms. This is apparent from the comparatively low dissolved-metal concentration found in the pore water surrounding the sediments.

Infauna residing within seafloor sediments also serve as indicators of marine contamination because of their limited mobility and well-defined responses to pollution. Numbers of species, abundance, biomass and other parameters describing infaunal community structure can indicate the presence of contaminant-induced stresses if, for example, gradients extending from a pollutant source to distant, unaffected areas are observed. More than 142,000 infaunal organisms have been collected and examined since the beginning of the benthic monitoring program two decades ago. Throughout the monitoring program, there has never been any indication of outfall-related impacts to benthic biota. Instead, the data have revealed a consistently healthy indigenous infaunal community, with uniformly high diversity that does not decline with proximity to the diffuser. These observations hold true despite widespread temporal variation in the abundance of individual organisms. Over the years, different taxa have dominated the infaunal community and then declined in response to natural changes in the environment. For instance, changes brought about by the strong El Niño in late 1997 and early 1998 resulted in one of the most diverse infaunal communities observed in more than a decade. El Niño was also responsible for a huge influx of juvenile sand dollars in 1999, mirroring a similar population increase in 1988 following the 1987 El Niño event. The infaunal trophic index provides an independent measure of the overall health of the benthic community. It remained uniformly high during 2005, as it has over the past nineteen years, reflecting a marine community dominated by suspension-feeding organisms living in clean sediments.

1.0 INTRODUCTION

The City of Morro Bay and the Cayucos Sanitary District jointly own the wastewater treatment plant operated by the City of Morro Bay. A National Pollutant Discharge Elimination System (NPDES) permit, modifying secondary treatment requirements, was issued to the City of Morro Bay and the Cayucos Sanitary District (MBCSD) in March 1985 (Permit Number CA0047881). The permit was issued by Region IX of the U.S. Environmental Protection Agency (USEPA) and the Central Coast California Regional Water Quality Control Board (RWQCB). The original permit expired in March of 1990, and, following an evaluation process, was reissued in March of 1993 (RWQCB-USEPA 1993ab). That permit expired early in 1998, but received an administrative extension through 11 December 1998 (RWQCB 1998). In December 1998, a new 301(h)-modified permit was issued, and became effective in early 1999 (RWQCB-USEPA 1998ab). This report addresses compliance with the provisions of the latest permit.

Marine Research Specialists began conducting the Offshore Monitoring and Reporting Program for the City of Morro Bay/Cayucos Sanitary District in July 1993. Since that time, samples have been collected and analyzed by the Treatment Plant staff, Marine Research Specialists, and subcontracted laboratories that included BC Laboratories and Aquatic Testing Laboratories in 2005.

1.1 REPORT ORGANIZATION

This year-2005 annual report summarizes results from the four major monitoring components that analyze the treatment plant, receiving waters, marine sediments, and benthic biota. The document is organized in a manner similar to that of previous annual reports to provide continuity in the reporting aspects of the monitoring program. Other than this brief introductory chapter, the major sections include the following:

Chapter 2	Treatment Plant Performance
Chapter 3	Receiving Water Quality
Chapter 4	Bottom Sediments and Biota
Chapter 5	Conclusions and Recommendations
Chapter 6	Literature Cited

Chapter 2 details all aspects of the onshore part of the monitoring program, including the operation and performance of the treatment plant, as well as a discussion of the collection and conveyance system as it pertains to the treatment process. It also describes sludge quality, and the disposition of biosolids during 2005. Chapter 3 documents the four offshore surveys that were conducted during 2005 to assess the quality of ocean waters where the effluent is discharged. Chapter 4 documents the analysis of sediment samples collected during an offshore survey conducted in October 2005. It places the chemical and biological measurements in historical context by incorporating time-series analyses of the entire 20-year database. Finally, Chapter 5 briefly reiterates the major conclusions of this report, and presents some recommendations for future actions concerning the operation of the treatment plant and the monitoring program.

In addition to these chapters, a large set of appendices provides detailed supporting documentation for the analyses contained in the body of this report. Appendix A lists the design specifications of the wastewater treatment plant. Quality control and quality assurance procedures for the analytical laboratories are provided in Appendix B. The results of all major chemical analysis performed on effluent and biosolids samples collected during 2005 are tabulated in Appendix C. The NPDES permit no longer requires separate reporting of data collected during the October benthic survey. Instead, they are included in Appendices D and E of this report. Appendix D documents physicochemical analyses performed on offshore benthic samples, and Appendix E details the biological analysis of offshore sediment samples.

CHAPTER 2

Treatment Plant Performance

2.0 TREATMENT PLANT PERFORMANCE

During 2005 as in prior years, the treatment plant efficiently removed a large fraction of organics and other solids from the wastewater stream. The dramatic difference between influent and effluent contaminant loading attested to the plant's high operational performance throughout the year. Because of a proactive operations and maintenance program, there were no exceptions to the limits specified in the NPDES discharge permit during 2005. The high operational performance of the plant is demonstrated by analyses of key diagnostic constituents, including suspended solids, biochemical oxygen demand (BOD), and oil and grease (O&G). The benign environmental character of the effluent is demonstrated through periodic analyses of effluent for trace metals, pesticides, priority pollutants, and toxicity. The sections below use these analyses to evaluate regulatory compliance through a comparison of measured wastewater characteristics and the limits cited in the NPDES discharge permit issued to the City of Morro Bay and Cayucos Sanitary District in December 1998 (RWQCB-USEPA 1998a).

Data collected during 2005 augment 19 years of prior monitoring information. This extensive database discloses a treatment process that has consistently performed at a high level for two decades. As in prior years, removal of suspended solids and oxygen-demanding materials during 2005 was consistently and substantially above permit-required minimum limits. The treatment plant's performance routinely approached and exceeded the requirements for full secondary treatment even though those requirements do not apply to this partial-secondary facility. In fact, the effluent usually consists entirely of wastewater that has received secondary treatment. Blending of primary- and secondary-treated wastewater is usually only required during periods of particularly high flow, such as during rainstorms. During 2005, the blending valve was open only 7.5% of the time.

Despite processing a high proportion of sewage during 2005, the plant discharged only 21% of the allowed solids, while still attaining removal rates that were 18% higher than the permitted minimum. Throughout the year, the treatment plant removed solids at a rate exceeding the 85% requirement for full secondary treatment. Average effluent suspended solids concentrations in eleven months were at or below the 30-mg/L criteria for full secondary treatment. During the other month (July), average concentrations were close to the standard, remaining below 35 mg/L. Settleable solids were imperceptible within the effluent samples collected throughout the year.

During 2005, the treatment plant also achieved a high removal rate for BOD (83%). The lowest monthly removal rate of 76.6% was more than double the minimum required rate of 30%. Average BOD removal rates for the remaining months met or exceeded 79% and, during four months, exceeded the 85% monthly removal standard for full secondary treatment. The average effluent BOD concentration (50 mg/L) was less than half the permitted maximum (120 mg/L), and the mass emission (88 MT) was less than a third of the allowed BOD discharge (342 MT).

The general absence of industrial contaminants in the wastestream bears testimony to the benign nature of the influent, which is almost entirely generated by non-industrial residential sources. Chemical analyses for 13 trace metals and over 150 synthetic organic compounds detected only a few low-level concentrations of common wastewater constituents, and five naturally occurring trace metals. All measured concentrations were well below permit limitations. Chronic bioassays of composite samples demonstrated that the effluent discharged during 2005 had low toxicity to marine organisms, as has been the case for the last decade. The acute toxicity bioassays conducted in 2005 were confounded by ammonia interference, tions that were slightly higher than the norm. None of the events during 2005 caused an exceedance of any of the NPDES permit limits.

2.1 Wastewater Treatment Plant

The Morro Bay/Cayucos Wastewater Treatment Plant (hereinafter WWTP) is publicly owned and operated under the auspices of the City of Morro Bay and the Cayucos Sanitary District (hereinafter MBCSD). The WWTP is located in the City of Morro Bay along the central coast of California within San Luis Obispo County (Figure 2.1). Based on the year-2000 census, the plant serves the communities of Morro Bay and Cayucos, which have a combined population of approximately 13,293, 10,350 for Morro Bay and 2,943 for Cayucos. The treatment plant discharged an average of 1.25 million gallons per day (MGD) during 2005. The plant was designed to accommodate an average dry-weather flow of 2.06 MGD, a peak seasonal dry-weather flow (PSDWF) of 2.36 MGD, and a maximum wet-weather flow (PWWF) of 6.64 MGD.

2.1.1 Operations

During 2005, eight trained personnel operated the Wastewater Treatment Plant (Table 2.1). Mr. Bruce Keogh was both the Wastewater Division Manager and Laboratory Director. Mr. Les Girvin continued as Wastewater Treatment Plant Supervisor. Mr. Eric Zatt obtained his Grade II certification on 16 April 2005.

Table 2.1 Morro Bay/Cayucos WWTP Personnel During 2005

Name	Grade and Certification No.
Bruce Keogh	IV – 8978
Les Girvin	III – 8499
Steven Aschenbrenner	II – 7548
Jim Hayes	II – 9221
David Williams	II – 9601
John Gunderlock	II – 10500
Eric Zatt	II – 10621
Steve Sibley	I – 8005

The WWTP personnel provided laboratory workspace to the National Estuary Program (NEP), which is dedicated to protecting and restoring the natural resources of Morro Bay and its watershed. NEP volunteers used the WWTP laboratory to analyze bacterial samples that were collected throughout the Morro Bay watershed.

WWTP personnel maintained their expertise by attending a number of workshops, seminars, and continuing-education classes throughout 2005. Training-course topics included confined space entry, CPR and first aid, driver awareness training, forklift operation, discrimination and harassment prevention, California biosolids regulations, the California Integrated Water Quality System (CIWQS), smoke testing methods, chlorine analysis, and supervisory skills training. WWTP personnel also conducted a number of plant tours for the public, students from local schools and colleges, equipment suppliers, and RWQCB staff.

In addition to performing the periodic effluent analyses required by the NPDES Permit, laboratories involved in the analyses of the MBCSD WWTP samples participated in laboratory performance evaluations. During 2005, the WWTP laboratory, and the laboratories subcontracted to the WWTP, conducted mandatory tests that are intended to evaluate the accuracy of effluent measurements and assure the overall quality of the monitoring reports. The WWTP laboratory analyzes effluent parameters that are reported on a daily and weekly basis, including residual chlorine, suspended solids, turbidity, coliform, pH, biochemical oxygen demand, and oil and grease. Aquatic Testing Laboratories conducts bioassay tests to evaluate the toxicity of effluent to marine organisms. BC Laboratories conducts semiannual chemical analyses to determine the concentration of a wide range of constituents in effluent, while Creek Environmental Laboratories performed the analysis of biosolid samples collected in 2005. Each laboratory's quality-assurance and quality-control plan is included in Appendix B of this report.

Several State and Federally sponsored laboratory performance evaluations were successfully completed during 2005. The results from all of these studies were deemed acceptable by the regulatory agencies. They included:

- The California Department of Health Services Environmental Laboratory Accreditation Program (ELAP) review of general laboratory protocols
- The California Department of Health Services Water Pollution Study-122 (WP-122) evaluation of microbial analysis capabilities,
- The United States Environmental Protection Agency's (USEPA) Discharge Monitoring Report-Quality Assurance Study (DMR-QA-25), which assessed the chemical analysis capabilities, and
- The USEPA DMR-QA Whole Effluent Toxicity Test (WETT) determination of the accuracy of bioassay capabilities.

In recent years, both the City of Morro Bay and the Cayucos Sanitary District have more actively sought to identify and eliminate sources of inflow (rainwater runoff) and infiltration (of groundwater) within their respective collection systems. In late 2000, a major source of groundwater infiltration was stopped after the 18-inch sewer line along Atascadero Road was repaired using cured-in-place pipe-lining technology. These and other repair projects have eliminated a substantial volume of groundwater infiltration into the collection system. These reductions in inflow and infiltration (I&I) have measurably decreased the amount of non-wastewater throughput that was unnecessarily processed by the treatment plant.

As part of the Atascadero Road project, the City of Morro Bay identified considerable damage in a major sewer lateral serving Morro Bay High School. This leaking lateral was replaced in 2001. Additional laterals that flow into the Atascadero Road sewer line were sealed and grouted to reduce leakage at the point where they connect to the new lining. At that time, pipe conditions in other areas were evaluated and prioritized for future repair. This process was continued during 2005 with the inspection, pressure washing, and repair, as needed, of all manholes encountered during the regular course of line cleaning, and the replacement of five manhole collars.

In addition to these measures, smoke testing was performed in the Northeast quadrant of the City. Smoke testing is an important tool in infrastructure management used to identify breaks, defects, and illegal connections to the wastewater collection system. By forcing smoke-filled air through a sewer or plumbing system, leaks can be quickly detected as smoke escapes through problem areas. Identifying and reducing the number of breaks, defects, and illegal connections helps to reduce I&I in the collection system.

During 2005, the City's Public Works Department also continued preformed cleaning and grit-removal within the City's collection system. Over the course of the year, a total of 312,053 feet of sewer line were cleaned. The program centers on a state-of-the-art hydrovac sewer cleaner purchased in 2001. This powerful hydrovac truck is capable of removing a large volume of grit from sewer lines that would normally be washed down the line during standard cleaning operations. Grit removal reduces the potential for line blockage and spills, and decreases the volume of solids that eventually enter the treatment plant and needlessly abrade the plant's equipment.

Additionally, in 2003 the City approved the commissioning of a Sewer Master Plan update. The updated Sewer Master Plan is currently in a draft form, and is slated for review and adoption by the City Council in 2006. The updated plan included the installation and monitoring of wastewater flow at eleven drainage basins within the City, which was used to quantify flow during significant rainfall events, address I&I, and evaluate the performance of individual lift stations. The plan also includes collection-system hydraulic modeling. Finally, it provides recommendations for a capital improvements program, including staffing and O&M budget analysis. During 2005, in anticipation of plan implementation, pump flow rates were analyzed, a new pump and a rebuilt pump were installed at Lift Stations #2 and #3, and the dry wells at both of these stations were patched, sealed and painted. Extra backup pumps at both Stations were also rebuilt, as were the check valves at lift Station #3. Finally, a Hatch Net fall-through prevention system was installed at lift station #1.

During 2005, the Cayucos Sanitary District (CSD) also continued to upgrade their collection system and proactively address potential areas of concern. Funding for major upgrades to the collection system originates in the District's five-year capital improvements program. This program is re-budgeted on an annual basis. During 2005, there were several significant improvements to the system beyond the more routine efforts to further reduce storm-drain cross-connections, seal visible leaks in manholes and chimneys, and install plastic inserts in manholes subject to surface water immersion.

For example, extensive reconstruction and relocation of Lift Station #2 at 24th Street and Pacific Avenue was completed in 2005. This project was initiated in July 2004 and included the replacement of the force main under the nearby creek, and the placement of a total of 843 feet of 16-inch-diameter HDPE pipe as a sleeve over an existing 10-inch-diameter HDPE pipe. A number of other major improvements to the CSD collection system were also initiated or completed during 2005. Damaged sewer lines at D St. and Ash Ave., and along Pacific Street were repaired. On Chaney Ave., 980 ft of damaged sewer line were rehabilitated, along with the sewer manhole. Similarly, throughout the District, a total of over 67,000 ft of sewer line were cleaned, and an additional 55,000 ft of line were inspected using video reconnaissance. Nine sewer manholes and 6 sewer cleanouts were also raised to grade. The digitization of District sewer easements and the creation of links to scanned plans were also performed.

At the WWTP, plant personnel continued an diligent, proactive preventative maintenance program that included repair of various components of the treatment process throughout 2005. These activities were documented in the monthly operations summaries submitted to the RWQCB (MBCSD 2005a-l). As part of the routine maintenance procedures, the exterior of the ocean outfall and the diffuser structure were inspected for signs of damage. The visual inspection was conducted by SCUBA on 13 October 2005. The outfall was found to be in good condition, with no broken or plugged diffuser ports. A video recording of the diffuser structure documented the continued presence of sensitive filter-feeding anemones and other high-relief hard-substrate organisms residing on and near the diffuser ports.

Figure 2.2 is a photograph of a diffuser port taken during a prior outfall inspection. It depicts a dense cover of marine epifaunal organisms thriving on the outer surface of the diffuser port. A large colony of club-tipped anemones (*Corynactis californica*), which are bright pinkish-red in color, covers the top surface of the port. Their presence attests to the benign nature of the effluent discharge, as well as to the value of the outfall as an artificial reef. Based on quantitative biological surveys conducted within the region, these anemones are more typical of high-relief rock surfaces that are only occasionally found within Estero Bay, and then only in deeper water (>85 m) (Morro Group, Inc. 1999). Ostensibly, their rarity on lower-relief rocks found nearshore arises from their susceptibility to elevated suspended sediment loads. Their preponderance on the diffuser confirms that the outfall's particulate discharge has little negative impact on marine organisms. In fact, the effluent's organic content probably provides a beneficial food source for these filter-feeding organisms.

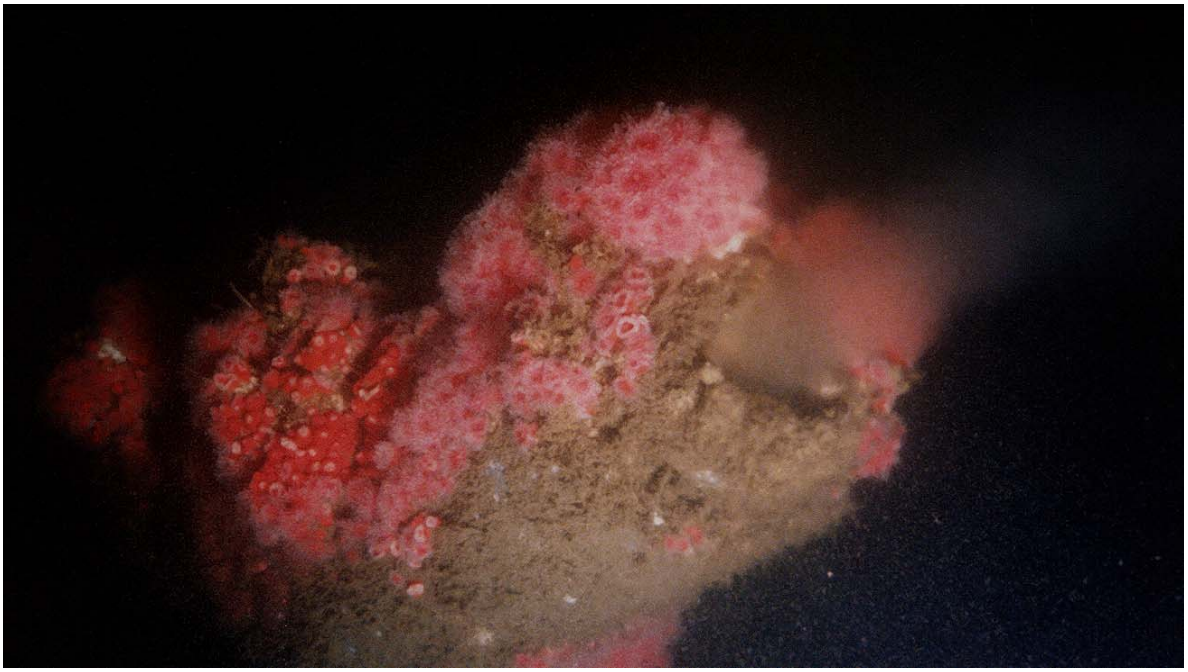

Figure 2.2 Photograph of a Diffuser Port Discharging Effluent

The two spar buoys marking the diffuser structure were also found to be in good condition during the visual inspection conducted on 13 October 2005. In 2003, the spar buoys were brought to the plant for repair and maintenance, where MBSCD staff removed marine growth, installed new sacrificial anodes, and provided new ground tackle before reinstalling the buoys.

Other routine maintenance and minor repairs were regularly conducted on various treatment-plant components. For example, the emergency standby generator was tested on a regular basis under full load. Every quarter, the standby generator was inspected, and a certified contractor performed scheduled maintenance. During several power outages experienced during 2005, it was found to be operating properly. Annual inspections were also performed on the Plant's cathodic protection system and back-flow prevention devices.

In addition, a number of major repairs and preventative-maintenance measures were instituted during 2005. The largest maintenance effort involved the preparations for taking (primary) digester #3 off-line for cleaning, inspection, and maintenance. Although the actual cleaning of the digester would not take place until January 2006, the preparations began in early 2005 with the conversion of digester #2 from a secondary to a primary digester. This slow process begins with heating and mixing of the digester. The second step in the process is to begin feeding the digester primary sludge, and then slowly increasing the amount of primary sludge feed to the digester. The process of converting the digester from a secondary to a primary digester requires several months, and was completed in May 2005. The timing of the project allowed the plant to operated two primary digesters during the summer months, thereby providing greater flexibility in plant operation when loading was heaviest.

The second half of the project began in September of 2005 with the reduction in the sludge feed to digester #3. As the biological processes within the digester slowed, staff stopped the digester gas recirculation mixing process. In October, three dewatering wells were installed to control groundwater levels once the digester was emptied. Over the next two months settled sludge was removed to the drying beds in preparation for the cleaning process. Then, on 28 December, staff began pumping nitrogen gas into the gas space inside the digester to purge methane gas from within the floating dome of the digester. This procedure minimizes the possibility of explosion when the digester is opened to the atmosphere.

The second major maintenance effort during 2005 involved the draining and inspection of both primary clarifiers during the month of November. On separate days, the clarifiers were taken off-line and drained. While off-line, the tanks and other essential components were inspected. Although no major problems were noted, a small quantity of rags was removed from primary clarifier #2. The clarifiers were off-line for approximately 24 hours each.

In addition, WWTP staff performed the following noteworthy maintenance activities in 2005:

- Installed alarm floats in both primary clarifiers. The alarm floats are tied into the existing auto-dialer alarm system. (January);
- Installed a new electromagnetic flow meter on the primary sludge pumping line to the digesters and replaced above ground piping with stainless steel piping. (January);
- Installed new vacuum and pressure relief valves on digesters #1, #2, and #3. The new valves replaced existing valves that are no longer cost effective to repair (January);
- Installed a new airflow fan and motor, and a new airflow switch on Boiler #2 (January);
- Reconfigured the piping of the sump pump system in the sodium hypochlorite containment area to allow staff or emergency responders to safely pump liquid from the area in the event of an accidental release from the chemical storage tank. (February);
- Repaired a fault with the control wiring on secondary effluent pump #2 at the interstage pump station (February);
- Installed a digester gas recirculation blower on digester #2 to improve the mixing process, and installed new piping on the digester gas recirculation system for digester #2 (March);
- Installed a new sample supply pump and a new sample cell assembly on the total chlorine residual analyzer. Staff also assembled a spare sample supply pump and sample cell assembly in order to minimize time off-line in the event of a mechanical malfunction of parts in the primary cell assembly (April);
- Replaced a faulty diaphragm on sodium bisulfite pump #3, and a faulty antisiphon valve on sodium bisulfite pump #2 (April);
- Replaced a faulty antisiphon valve on the discharge piping of sodium bisulfate pump #1 and a faulty sump pump used to supply effluent sample to the chlorine residual analyzer (May);
- Installed a new compressor head on instrument air compressor unit #9312 (May);
- Rebuilt the main influent pump (#1). Installed new seals and bearings and new U-joints on the pump drive shaft (June);
- Installed a new motor on the chlorine contact drive unit that powers the flights in the chlorine contact chamber (June);
- Replaced a faulty flow-paced sodium bisulfate-dosing pump. Following removal, the pump was rebuilt to serve as a backup pump (July);

- Relocated the twenty-four hour effluent composite sampler to the air relief structure so that grab and composite samples are collected proximally (July);
- Installed new pump on the secondary clarifier center column sprayer and scum-trough wash-down system (August);
- Replaced sodium hypochlorite dosing pump #2 with a new pump (September);
- Re-installed main influent pump #3, which had been taken off-line for inspection and maintenance. Rebuilt the pump, installed new seals and bearings, and installed new U-joints on the pump drive shaft (September);
- Installed a new Chemineer Model 1 GTD Flashmixer in the chlorine contact tank to replace an existing unit whose repair was no longer cost effective (October);
- Replaced the existing 4" steel pipe air manifold in the mixed liquor solids contact channel with a new 4" PVC manifold. The new piping should increase the efficiency of the aeration system in the mixed liquor channel (November);
- Cleared a blockage in the bubbler tube at the headworks which was providing faulty level readings in the headworks influent channel resulting in erratic operation of the influent pumps (December); and
- Replaced a faulty alarm float on primary clarifier #2 (December).

During 2005, the MBCSD also continued to sponsor a pilot study testing the feasibility of windrow composting to improve the quality of biosolids produced by the WWTP. Changing the current Class-B biosolids designation to Class-A (exceptional quality, composted) biosolids will allow the final product to be made available to the public for beneficial reuse as a high-quality soil amendment.

Upgrades to the plant planned for the future include the institution of a co-generation project that will harvest the methane produced from the digesters. The feasibility of the project has been thoroughly investigated, and in August 2004, the MBCSD entered into a contract with US Energy to use the methane produced from this venture. Throughout 2005, plant staff worked with US Energy to formulate the design and construction of the system.

2.1.2 Plant History

The original WWTP was built in 1954 and was enlarged 10 years later to meet the demands of a growing coastal community. Its initial capacity was 0.7 MGD with a 1-MGD maximum. The original plant included a headworks structure, primary and secondary clarifiers, a biofilter, a single-stage digester, chlorination facilities, biosolids drying beds, and a short ocean outfall.

The plant was upgraded in 1964 to a nominal capacity of 1 MGD and a 1.3-MGD maximum. This upgrade added a pump station, a splitter box, a primary clarifier, a secondary clarifier, a biofilter, chlorination facilities, biosolids beds, and another primary digester, which allowed the existing digester to be converted to a secondary capacity. A new office and laboratory were also constructed during this upgrade.

During the 1970s, the City of Morro Bay developed a plan for additional upgrades to the existing facilities to further augment the plant's capacity. It included the construction of a new outfall to better protect the marine environment. The new outfall was designed following a yearlong study of oceanographic conditions within Estero Bay. The new outfall and diffuser system extended the discharge from the surfzone to a point well offshore, in deeper water. Deeper discharge provides increased dilution of wastewater within an open-ocean environment.

Design of the planned facility improvements commenced in 1980. The design of the new outfall was completed in April 1981. In June 1982, immediately after construction was completed, the new outfall was placed into service. The design of the treatment-plant improvements was completed in September 1981. The design called for a final effluent suspended-solids concentration of 30 mg/L and an equivalent BOD load. However, financial aid from state or federal agencies to finance the construction of a treatment plant that could meet these full secondary-treatment levels was not available at that time. Because discharge through the new outfall from the old plant was not causing any apparent adverse environmental impact, and because the projected flow rates were low, the state could not justify providing financial aid for the plant upgrade. At that time, other higher-priority projects were seeking similar financial assistance. As a compromise, the treatment plant's design was modified to provide secondary treatment to a majority (1 MGD), but not all, of the projected flow. In this way, the state water-quality standards, as set forth in the California Ocean Plan or the COP (SWRCB 1990), could still be met. State officials concurred with this level of treatment, provided that the USEPA approved a 301(h)-modified NPDES discharge permit that adjusted full secondary-treatment requirements on suspended-solid and BOD emissions.

In a letter dated 16 March 1983, the staff of the California Regional Water Quality Control Board (RWQCB) indicated their tentative determination that the proposed altered discharge would comply with applicable state laws, including water-quality standards, and would not result in requirements for additional treatment, pollution control, or other requirements on any other point or non-point source. This tentative determination was made pursuant to Title 40 of the Code of Federal Regulations, Section 125.60(b)(2) [40 CFR 125.60(b)(2)] (USGPO 1982a) and 40 CFR 125.63(b) of the 301(h) regulations dated November 1982 (USGPO 1982b).

Treatment-plant upgrades performed between 1983 and 1985 increased plant capacity to 2.06-MGD average dry-weather flow and a peak flow of 6.6 MGD. The plant now includes primary treatment of all influent by screening, grit removal, and primary sedimentation. Depending on the hydraulic conditions within the plant, the majority of the flow (up to 1 MGD) can be diverted through a secondary treatment process consisting of trickling filters, clarifiers, and a solids contact chamber. This achieves at least a 75% solids removal in the subsequent blend of primary and secondary effluent. The new secondary-treatment process utilized the two existing trickling filters, a new aerated solids contact channel, and a new secondary sedimentation tank. The original 1954 sedimentation tank was converted into a chlorination system in which the primary and secondary effluents are mixed and disinfected prior to dechlorination and discharge through the ocean outfall.

On 29 March 1985, water-quality standards for the MBCSD treatment plant were established in an NPDES permit based on a Section 301(h) modification. The permit mandated that the treatment plant achieve a suspended solids content of no more than 70 mg/L (75% removal) and a maximum BOD of 120 mg/L. An extensive monitoring program was also required to assure the maintenance of environmental quality. The permit was valid for five years and expired on 8 March 1990. Following an evaluation process, the permit was reissued in December 1992. Improvements to the treatment facilities during this period included the installation of a sludge removal system in the mixing chamber of the chlorine contact tank. Sludge buildup in the mixing chamber had been increasing the effluent BOD, so a system of high-pressure nozzles and scraper flights was installed to alleviate this problem. Effluent BOD declined after this installation.

During the permit renewal process in 1992, concerns were expressed that the discharge could affect the waters within Morro Bay, to the south of the discharge. Consequently, a shellfish-monitoring program was instituted. Subsequent analysis demonstrated that the discharge was too far from this newly designated national marine estuary to materially affect its waters. At a typical 1-cm/sec flow speed, effluent would take more than three days to reach the entrance to the Bay located 2.8 km (1¾) miles to the south. Over a much shorter distance and time, intense open-ocean turbulence rapidly mixes and disperses the effluent beyond recognition. This is evident from the inability of extremely sensitive instruments to detect the presence of the effluent plume even short distances beyond the zone-of-initial dilution, which extends only 15 m from the diffuser structure (see Chapter 3). Similarly, effluent discharge does not affect the Monterey Bay National Marine Sanctuary, whose southern boundary is near Cambria Rock, 20 miles to the north.

In anticipation of permit expiration in early 1998, the MBCSD applied for renewal of their 301(h)-modified NPDES permit in May 1997. The application was supported by an extensive technical review of more than a decade of monitoring data (MRS 1997a). An administrative extension was granted through 11 December 1998 to allow time for review and issuance of a new discharge permit (RWQCB 1998). A series of discussions with RWQCB and USEPA staff concerning new permit revisions were held throughout 1998. Changes implemented in the new permit included:

- a 12.7% reduction in the allowed mass-emission of suspended solids, BOD, and Oil and Grease,
- more extensive reporting requirements for biosolids,
- elimination of shellfish monitoring,
- a revised benthic sampling pattern eliminating seasonal sampling and increasing the number of stations close to the diffuser structure,
- a revised receiving-water sampling program eliminating bottle casts and doubling the number of CTD casts close to the diffuser structure, and
- specification of mass emission goals for toxic chemicals.

In July 1998, RWQCB staff determined that the discharge described in the MBCSD application '…*would comply with applicable state laws, including water quality standards, and would not result in additional treatment, pollution control, or other requirements on any other point or nonpoint source.*' In September 1998, Region IX of the USEPA issued a tentative decision that MBCSD be granted a permit that modifies secondary treatment requirements under Section 301(h) of the Clean Water Act. In December 1998, the RWQCB approved a modified NPDES permit that moderated some of the full secondary treatment requirements (Permit No. CA0047881). On 13 January 1999, the California Coastal Commission ruled that the Waste Discharge Requirements incorporated in the current NPDES permit were consistent with the California Coastal Zone Management Act. On 26 January 1999, the USEPA issued the latest NPDES permit, effective 1 March 1999.

The current NPDES permit was approved by the RWQCB with the proviso that a comprehensive water-reclamation feasibility study be conducted within the following year (that is, in 1999). A draft report was submitted to the RWQCB on 7 October 1999 that included a detailed evaluation of reclamation options and feasibility, and a review of water supply and demand for the region around Morro Bay and Cayucos (Carollo Engineers 1999). Staff of the RWQCB provided comments on the feasibility study on 16 February 2000, and the report was finalized after these comments were included by reference.

The report considered water reuse for irrigation of agricultural crops, irrigation of landscapes, parks and public facilities, industrial reuse, stream-flow augmentation, estuary/wetlands enhancement, groundwater recharge, and seawater intrusion barriers. The study found that implementation of a reuse project would require upgrading the entire wastewater flow to full secondary treatment, and that the portion of wastewater going to reuse would need to be upgraded to tertiary treatment. The study also determined that the existing water supply adequately meets demand, and there would be little utilization of recycled water.

The current NPDES permit was due to expire on 1 March 2004. In anticipation of its expiration, an application for a new discharge permit was submitted on 3 July 2003 (MBCSD 2003). The application included a supplemental report (MRS 2003a) that provided an extensive technical review of monitoring data collected over the previous decade and a half. Based on the general absence of perceptible impacts from the historical discharge, and high effluent quality projected for the upcoming five-year permit term, an application was submitted based on a continuation of discharge under a 301(h) modification of secondary treatment requirements. On 4 February 2004, the RWQCB (2004) administratively extended the current permit.

2.1.3 Regulatory Setting

The 1972 Federal Clean Water Act and its amendments in 1977 established national water-quality goals and created a national permit system (NPDES) with minimum standards for the quality of discharged waters (USGPO 1997a). It required states to establish standards specific to water bodies and designated the types of pollutants to be regulated. Since 1973, the California State Water Resources Control Board and its nine Regional Water Quality Control Boards have been delegated the responsibility for administering permitted discharges into the coastal marine waters of California. The State Board prepares and adopts the COP, which incorporates the state water-quality standards that apply to all NPDES permits. The Central Coast Region of the California Water Quality Board (RWQCB) established a Water Quality Control Plan for the basin containing San Luis Obispo County waters (RWQCB 1994). The basin standards incorporate the applicable portions of the COP, and are more specific to the beneficial uses of marine waters adjacent to the outfall site. Water-quality objectives and toxic material limitations in the basin plan are designed to protect the beneficial uses of ocean waters within specific coastal areas. The basin plan identifies the following existing beneficial uses for the waters of Estero Bay.

- ***Water Contact Recreation (REC-1).*** *Uses of water for recreational activities involving body contact with water, where ingestion of water is reasonably possible. These uses include, but are not limited to, swimming, wading, water skiing, skin and scuba diving, surfing and fishing.*

- ***Non-Contact Water Recreation (REC-2).*** *Uses of water for recreational activities involving proximity to water, but not normally involving body contact with water, where ingestion of water is reasonably possible. These uses include, but are not limited to, picnicking, sunbathing, hiking, beachcombing, camping, boating, tidepool and marine life study, hunting, sightseeing, and aesthetic enjoyment in conjunction with the above activities.*

- ***Industrial Service Supply (IND).*** *Uses of water for industrial activities that do not depend primarily on water quality, including, mining, cooling water supply, hydraulic conveyance, gravel washing, fire protection, or oil well repressurization.*

- ***Navigation (NAV).*** *Uses of water for shipping, travel, or other transportation by private, military, or commercial vessels. The regional water quality control board interprets NAV as any natural body of water that that has sufficient capacity to float watercraft for the purposes of commerce, trade, transportation, and pleasure.*

- ***Marine Habitat (MAR).*** *Uses of water that support marine ecosystems including, but not limited to, preservation or enhancement of marine habitats, vegetation such as kelp, fish, shellfish, or wildlife, such as marine mammals and shorebirds.*

- ***Shellfish Harvesting (SHELL).*** *Uses of water that support habitats suitable for the collection of filter-feeding shellfish such as clams, oysters, and mussels, for human consumption, commercial, or sport purposes. This includes waters that have in the past, or may in the future, contain significant shellfisheries.*

- ***Ocean Commercial and Sport Fishing (COMM).*** *Uses of water for commercial or recreational collection of fish, shellfish, or other organisms, including uses involving organisms intended for human consumption or bait purposes.*

- ***Preservation of Rare, Threatened, or Endangered Species (RARE).*** *Uses of water that support habitats necessary, at least in part, for the survival and successful maintenance of plant or animal species established under state or federal law as rare, threatened, or endangered.*

- ***Wildlife Habitat (WILD).*** *Uses of water that support terrestrial ecosystems including, but not limited to, preservation and enhancement of terrestrial habitats, vegetation, wildlife (e.g., mammals, birds, reptiles, amphibians, invertebrates), or wildlife water and food sources.*

Section 301(b) of the Clean Water Act requires publicly owned treatment works (POTW) to meet effluent limitations based on secondary treatment, which is defined in terms of limits on three technology-based effluent parameters (40 CFR 133; USGPO 1997a). These limitations are:

- Total suspended solids (TSS) concentrations not exceeding 30 mg/L as a 30-day average and removal rates not less than 85%;
- Biochemical oxygen demand (BOD) concentrations not exceeding 30 mg/L as a 30-day average and removal rates not less than 85%; and
- Hydrogen-ion concentration (pH) between 6.0 and 9.0.

Section 301(h) of the Act allows an NPDES permit to be issued that modifies some or all of these full secondary treatment requirements, if certain conditions are met. The MBCSD WWTP is a combined primary and secondary treatment facility that has operated under a Section 301(h)-modified NPDES permit (number CA0047881) since March 1985. The modification in this NPDES permit applies only to the TSS and BOD requirements, so all other NPDES limitations apply without exception, including those for wastewater pH and toxic compounds. The modification was issued only after additional conditions were satisfied by the MBCSD. These included the following:

- The existence of a water-quality standard specific to the pollutant for which the modification is requested (40 CFR 125.61; USGPO 1997a). The COP specifies limits on TSS and dissolved oxygen depression (SWRCB 1997). In January 1999, the California Coastal Commission determined that the discharge is consistent with the State Coastal Zone Program that incorporates standards specified in the Ocean Plan.

- The discharge does not adversely impact public water supplies or interfere with the protection and propagation of balanced, indigenous biological populations (40 CFR 125.62). The U.S. Fish and Wildlife Service (USDOI 1998) and the National Marine Fisheries Service (NOAA 1998 2004) have determined that the discharge will not adversely impact threatened or endangered species, or critical habitats, pursuant to the Endangered Species Act.

- A monitoring and reporting program is in place that is capable of evaluating the effects of the discharge (40 CFR 125.63).

- The discharge will not result in any additional treatment requirements on any other point or non-point source (40 CFR 125.64).

- Because there are no known sources of toxic pollutants or pesticides in the influent, the WWTP is exempt from general pretreatment requirements in lieu of a pollution prevention program. Because the discharge is considered small, it is exempt from the urban pretreatment requirement (40 CFR 125.65).

- The MBCSD pollution-prevention program implements public-education and source-reduction programs to limit the entrance of toxic pollutants or pesticides into the treatment plant. Therefore, it meets the requirement for a non-industrial source control program (40 CFR 125.66).

- There will be no new substantially increased discharge of BOD and TSS beyond those specified in the permit (40 CFR 125.67). This conclusion is based on the historically high performance of the plant and the limited projected growth in population and industry within the service area. The analyses in this report demonstrate the continued veracity of this conclusion during 2005.

- The WWTP exceeds the minimum requirements for primary treatment (40 CFR 125.60) because it performed: '*…treatment by screening, sedimentation, and skimming adequate to remove at least 30 percent of the biochemical oxygen demanding material and of the suspended solids in the treatment works influent, and disinfection, where appropriate*' (40 CFR 125.58(r); USGPO 1997a).

The MBCSD WWTP is categorized as a Class III wastewater treatment facility by the California State Water Resources Control Board, Office of Operator Certification. It was reclassified in 2001 from a Class-IV facility. The plant classification was changed based on the treatment process and volume of flow through the plant. A typical Class-IV facility treats in excess of 20 MGD in the primary process, while the MBCSD plant processes a total flow of approximately 1.1 MGD. The change in classification acknowledges the exceptionally small volume of flow handled by the plant, and reduces the operator grades required by individuals running the plant.

2.1.4 Description of the Treatment and Outfall System

The WWTP operating characteristics are listed in Appendix A. All wastewater is treated through a primary treatment process, which includes screening, grit removal, and primary sedimentation, as shown in Figure 2.3. Typically, the vast majority of the flow (often more than 1 MGD) is diverted for an additional secondary treatment process consisting of biofilters, a solids-contact chamber, and a secondary clarifier. The secondary process consists of parallel single-stage, high-rate, trickling filters whose combined outflow goes to a solids contact channel, and then to a secondary sedimentation tank. Secondary-treated effluent is subsequently blended with primary effluent. The entire blend is chlorinated for disinfection and then dechlorinated. The disinfected effluent is discharged into Estero Bay through a 4400-ft (1341-m) outfall terminating in a multi-port diffuser system. Waste biosolids are anaerobically digested, dried, composted and used as soil conditioner/fertilizer. A schematic of the biosolid process is shown in Figure 2.16 on Page 2-47.

The location of the Morro Bay-Cayucos WWTP and outfall within Estero Bay is shown in Figure 2.4. The discharge is into unstressed, open-ocean waters at 35°23'11"N latitude and 120°52'29"W longitude. Treated effluent flows through a 27-in (0.69-m)-diameter outfall that extends a distance of approximately 4,400 ft (1,341 m) in a northwesterly direction. The outfall terminates in a multi-port diffuser situated approximately 2,700 ft (827 m) from shore. The diffuser lies in 50 ft (15.2 m) of water, referenced to Mean Lower Low Water (MLLW). Twenty-eight of the 34 available diffuser ports are currently open. The remaining six ports can be made operational if the sustained discharge exceeds 6.60 MGD. The diffuser is 170-ft (51.8 m) long.

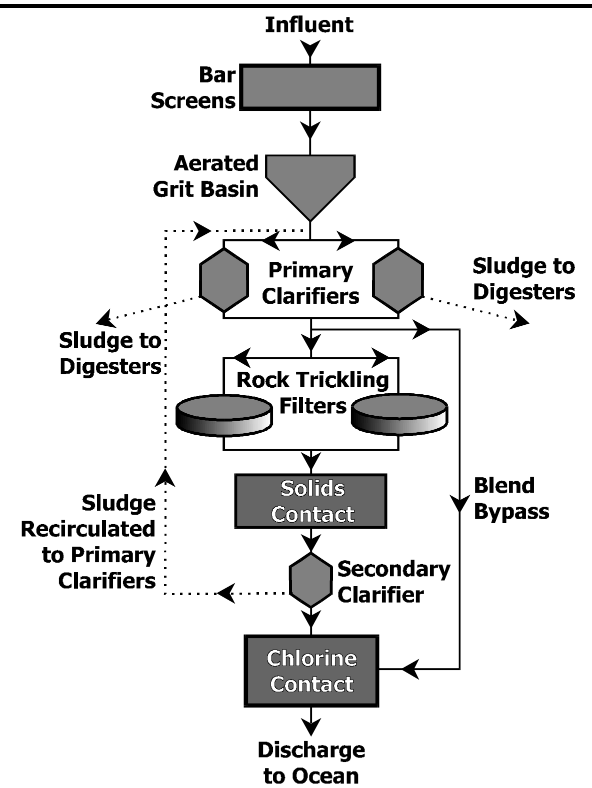

Figure 2.3 Schematic of the Wastewater Treatment Process

Figure 2.4 Location of the MBCSD Outfall and Monitoring Stations within Estero Bay

Because of its location, the MBCSD discharge does not interfere with the maintenance of water quality and designated beneficial uses within Estero Bay (see Section 2.1.3 on Page 2-11). The discharge occurs in well-flushed, open coastal waters where re-entrainment or accumulation of effluent will not result in violations of applicable water-quality standards, even if combined with pollutants from other sources. Intakes and outfalls from other publicly owned treatment works (POTWs) are far removed from the MBCSD outfall. For example, water intake for the desalinization plant is from salt-water wells and not from the ocean environment where the MBCSD discharge occurs. Similarly, surface discharge of water from the Duke Energy power plant (formerly PG&E), when it does occur, is located well south of the MBCSD discharge point and does not add significant chemical loads to the ocean environment.

2.1.5 Pollution Prevention Program

The MBCSD continued to advance its Pollution Prevention Program along several fronts during 2005. The program's purpose is to minimize the introduction of pollutants and pesticides into the treatment process. The NPDES permit requires an annual status report detailing efforts to comply with the requirements for a Pollution Prevention Program. This section serves as that report. Three aspects of pollution prevention were emphasized during 2005. First, the comprehensive industrial waste survey that was completed in 1999 has been updated annually, including in 2005. Second, a public outreach program, initiated in 1999, was continued in 2005. As the result of an vigorous advertising campaign over the past several years, the household hazardous waste facility was heavily used during 2005, with between 20 and 50 participants utilizing the facility each weekend. Finally, influent and effluent were diligently monitored for industrial contaminants.

Industrial Waste Survey

The MBCSD economy is predominately based on commercial fishing and tourism, with no heavy industry or manufacturing of environmental significance. As testimony to its tourism base, an unusually large number of restaurants, approximately 50, are listed in the industrial database for this small community. Accordingly, comprehensive monitoring for more than a decade and a half has found that effluent discharged from the treatment plant consists largely of relatively benign constituents. The industrial waste surveys conducted during 2005 and in prior years support these findings because they quantified a comparatively low volume of influent from light industrial sources containing nontoxic constituents typical of wastewater from domestic sources. Over all the years of modern treatment-plant operation, there has been virtually no expansion of the industrial base in this largely residential community. There is no input from heavy industrial sources. During 2005, the absence of significantly elevated levels of industrial pollutants within wastewater was indicative of the continued lack of substantive industrial input.

During 2005, industrial-waste survey activities relied heavily on the results of a detailed survey originally conducted in 1999. An update of the survey was completed in 2005. The design and implementation of a digital database included an inventory of all potential industrial users located within the service area. The database lists business names, addresses, names of contacts, telephone numbers, inventories of chemicals, discharge volumes, and other pertinent information. The computer database expedites quantitative assessments of industrial usage, and facilitates annual updates of contact information.

The master list of businesses in the database was developed from business license applications on file with the City of Morro Bay and from input provided by the Cayucos Sanitary District. The master list is subdivided into businesses that have no potential for industrial discharges, such as offices and retail stores, and those that have the potential for industrial waste discharge. All businesses with potential industrial discharge were either interviewed in person or provided with a Commercial User Questionnaire.

Based on the more-detailed information provided in the questionnaire, additional businesses were excluded from further consideration as industrial dischargers because they either did not generate wastewater at all, or discharged only domestic wastewater. These businesses included theaters, beauty shops, barbershops, and various retail firms. For the remaining industries, waste discharge volumes were estimated in proportion to water usage determined from billing records provided by the City Water Department. Follow-up activities for these businesses included scheduled return visits, surprise onsite inspections, and formal tours of the facilities. Based on this phase of the survey, certain industrial dischargers were identified for continued close monitoring by future site visits, scheduled annually. These include the commercial laundries, car washes, dry cleaners, print shops, and the oil-water separator maintained by the City of Morro Bay.

Overall water use within the communities has changed little between 1999 and 2005. The largest industry within the City of Morro Bay is the Duke Energy (formerly PG&E) steam power-generating plant when it is operating. However, none of its process flows into the sewer system. It discharges only domestic wastewater and small amounts of laboratory reagents into the wastestream processed by the treatment plant. Additionally, the Hazardous Chemical Business Plan filed by Duke Energy identifies all potentially toxic materials stored onsite and outlines plans for mitigating accidental spills, should they occur.

Based on data in the original 1999 Industrial Waste Survey and its subsequent updates, several businesses were targeted with unannounced inspections in 2005. They included: Mission Linen, Bayside Care Center/Casa de Flores, and Bayside Café.

Table 2.2 lists the 12 businesses with the greatest water usage as identified in the industrial-waste database. Businesses were subdivided into two groups depending on their type of discharge. The first group of businesses discharges wastewater that is similar to domestic waste from residential users. These businesses include the Bayside Care Center, Morro Bay High School, a hotel, and three public laundromats. Bayside Care Center is a senior care facility that has formerly been known as both Casa de Flores and Seashell's Communities. The Bayside Care Center houses approximately 220 residents and has dining and laundry facilities on site. The Bayside Care Center laundry facilities wash approximately 4,000 lbs of laundry in a 20-hour daily operating schedule. Their water use includes irrigation of the grounds, so not all of the 0.021 MGD ends up in the collection system.

Table 2.2 Businesses with the Largest Estimated Water Use

Business	Water Use (gpd)[1]	Fraction[2] (%)
Non-Industrial		
High School Agriculture	32,405	2.58
Bayside Care Center	21,149	1.69
Inn at Morro Bay	19,951	1.59
Morro Bay High School	3,104	0.25
Del Mar Laundromat	2,420	0.19
Morro Bay Launderland	2,015	0.16
Washing Well Laundromat	1,011	0.08
SubTotal	82,055	6.54
Light Industry		
Mission Linen	16,040	1.28
Morro Bay Mobile Station	5,074	0.40
Morro Bay Car Wash	3,476	0.28
Culligan	3,150	0.25
J&H Car Wash	569	0.05
SubTotal	28,309	2.26
Total Use	**110,364**	**8.80**

[1] Gallons per day [2] Fraction of average WWTP flow for 2005 (1.254 MGD)

Similarly, most of the 0.032-MGD water used by the agriculture program at Morro Bay High School is not discharged to the treatment plant. Their water usage doubled after the installation of football and baseball fields at the school but wastewater discharge remained relatively constant. Wastewater discharged by Morro Bay High School consists mostly of effluent similar to residential users, albeit in higher volume. Finally, the three public laundromats discharge wastewater that contains constituents similar to residential users, namely, detergents, but at volumes larger than their residential counterparts. Even so, total discharge from all of these non-industrial businesses constituted only 0.082 MGD or 6.5% of the influent processed by the treatment plant during 2005.

Dischargers of a more industrial nature include Mission Linen, the Culligan soft-water regeneration plant, and three car washes. In total, discharge from these light industries constitutes 2.3% (0.028 MGD) of the wastewater processed by the treatment plant. In addition, the constituents added by the businesses are not particularly toxic to humans or aquatic organisms, and are not usually thought to interfere with the operation of the treatment works. The only chemical Culligan adds to the wastestream is sodium chloride, which is of little significance to oceanic discharge. By far the largest contributor of wastewater is Mission Linen, at 0.016 MGD (1.3%). This commercial laundry uses several industrial-grade detergents, bleaches, surfactants, and brighteners. Some of these compounds can be harmful to bacteria within the secondary treatment system, and there is a potential for contaminants entering the wastestream from solvents, oils, and other substances that are washed from material that is laundered. Similarly, the three car washes were included on the list of light industrial users because of the volume of solids, oils, and greases that are washed from vehicles. All three car washes employ grease separators to pre-treat their wastewater prior to discharge into the collection system.

The survey results demonstrate that the volume of wastewater contributed by business users is negligible compared to domestic sources. Total input was approximately 8.8% (0.11 MGD) of plant flow during 2005. In addition, the character of their discharge was relatively benign in terms of the introduction of potential contaminants to the treatment-plant process. Smaller contributors, such as gas stations and repair garages, are restricted from disposal of known contaminants into the collection system by authority of the sewer-use ordinance within the City of Morro Bay municipal code. Similarly, under the municipal code, restaurants and self-service car washes are required to install and maintain grease traps within sewer line connections.

In addition to chemicals from the industrial sources that contribute to the influent, there are three chemicals purposefully introduced into the treatment process by the WWTP itself. Sodium hypochlorite is added to disinfect the wastewater prior to discharge through the ocean outfall. Currently, most wastewater facilities disinfect with some form of chlorine. A concern with chlorine disinfection is the possibility of a reaction between organic matter in the wastewater and residual chlorine. In addition, residual chlorine, even at low concentrations, can be hazardous to aquatic life. This means that treatment plants must dechlorinate to remove free and combined chlorine ions, thereby reducing effluent toxicity. Sodium bisulfite was used by the MBCSD as the dechlorinating agent during 2005. The amount of sodium hypochlorite used by the plant ranged from a minimum of 3,017 gal in October to a maximum of 4,204 gal in March (MBCSD 2005jc). Sodium bisulfite usage generally tracked chlorine usage, exhibiting a peak in usage of 2,930 gal in March (MBCSD 2005c). The third chemical used at the treatment plant was ferrous chloride. It was primarily used for odor control in the collection system and to reduce hydrogen sulfide emissions during flaring of digester gas and heating of the digesters at the WWTP. The latter application was required by the Air Pollution Control Board.

Public Outreach

Public awareness literature, which pertains to the proper disposal of hazardous household chemicals, is considered an integral part of the pollution prevention program. A public-services newsletter is included in water bills sent to residential users. A portion of this newsletter deals with proper waste-disposal techniques for constituents such as oil and grease and insecticides. Disposal guidelines are also discussed with industrial users during the industrial-waste survey site visits. Industrial users are told how strict adherence to the disposal guidelines minimizes damage to the treatment plant and collection systems, and how it ensures effective treatment of wastewater. The sewer-use ordinance within the City of Morro Bay municipal code also provides the legal authority for protection of the WWTP and collection systems. It limits the introduction of pollutants that could: a) interfere with the operation of the treatment works, b) pass through the WWTP into the environment, c) reduce the ability to reuse wastewater and biosolids, or d) expose WWTP employees to hazardous chemicals.

A major milestone in public outreach was achieved during 2000 when a new household hazardous waste disposal site was established at the wastewater treatment plant. Prior to that time, household waste disposal was only available on limited days at the distant Chicago Grade and Cold Canyon landfills. Treatment-plant staff assisted the Integrated Waste Management Authority in obtaining funding to establish a permanent collection facility on the grounds of the treatment plant. The facility opened to the public in August 2000. Since then, it has been open every Saturday from 11am to 3pm, except holiday weekends.

An incessant public-education campaign advertises the availability of the hazardous-waste disposal site and has been instrumental in making it one of the top sites in the county, in terms of the volume of material collected. The availability of hazardous waste disposal at the treatment plant, as well as general information about waste disposal, continued to be heavily advertised in the local media throughout 2005. Between 20 and 50 individuals used the household hazardous waste disposal facility each weekend, with approximately 1,500 people disposing of waste at the facility during the course of the year. From July 2004 to June 2005, over 73 tons of household hazardous wastes were processed through the disposal site, and a large portion of the collected material was recycled. Collected materials included over 28,413 pounds of oil-based paint, 20,102 pounds of latex paint, 7,063 pounds of flammable liquids and aerosols, 3,608 pounds of anti-freeze, 35,100 pounds of car batteries, 8,240 pounds of motor oil/oil products, and 850 pounds of household batteries. Without the permanent hazardous waste facility, much of this material would have ended up in a landfill or been discharged into the ocean.

Additionally, in 2005, the hazardous-waste disposal site collected 35,100 pounds of video display devices such as television and computer monitors, which the facility expanded to accept during 2004. The expansion helps to address a long-standing problem with the proper disposal of electronic waste, or e-waste as it is known. Cathode Ray Tubes (CRTs), the picture tubes used in many computer monitors and television screens, contain large quantities of five known toxic chemicals. Cadmium, mercury, chromium, lead, and phosphorous are all typically found in CRTs. In fact, the EPA estimates that discarded CRTs contribute at least 40% of the known lead in U.S. landfills. Flat panel monitors, cell phones, digital cameras, and other computer hardware also contain multiple hazardous waste elements. Although, normally, these contaminants are safely sealed within the finished product, the heavy equipment used to compact wastes at a landfill compromises the structural integrity of these products. Once the structural integrity of the housing is compromised, the hazardous materials are released into the surrounding area, where a leaching process occurs. Proper recycling of these products can prevent such contamination.

Source Identification

Although waste minimization measures have been implemented to reduce the introduction of incompatible contaminants into the wastestream, elevated concentrations are occasionally noted in the periodic chemical analysis of effluent. When this occurs, personnel from the City of Morro Bay and the Cayucos Sanitary District trace the contaminants to the source, and work with the source owner to eliminate the contamination. No unusual contaminants were detected within the wastestream during 2005. Some of the source-identification efforts conducted in recent years are described below.

Throughout 2003 and early 2004, rubber exam gloves and incontinence protection products were found to be causing fouling of the wet well at Lift Station #3. This prompted both an increased maintenance and cleaning schedule at this lift station, as well as multiple site visits to Casa de Flores/Bayside Care Center, a local nursing care facility. WWTP staff discussed the issue with the facility's Director of Environmental Services and paid several return visits over the course of the year to follow up on the situation. Maintenance concerns at the wet well of lift station #3 continued, however, into the spring of 2004. When compliance remained an issue, plant staff began to work with various City staff members, including the code enforcement officer, to review and develop a strategy for controlling the contents of the discharge from this facility. On 12 March 12 2004 an issuance of violation was issued by the City concerning the illegal discharge. On 26 April 2004, the Bayside Care/Casa de Flores staff and city staff met to formalize Bayside Care Center's plan of correction. As part of the plan of correction, on 25 May 2004, City staff provided in-service training to the Casa de Flores staff concerning the need to control the content of the discharge from their facility, and the results of improper disposal. Additional information was also provided to the Center's staff regarding the Clean Water Act and the estuary, and proper disposal of fats, oils, and grease by treatment plant personnel. In the wake of these efforts, Bayside Care Center implemented several improvements to their facility including installation of a new, larger grease trap in the kitchen and a new catch basin with two filters in the laundry, and a more thorough laundry sorting process by facility staff. Since the institution of these improvements, fouling of the wet well at Lift Station #3 has declined, and no problems were reported during a February 2005 visit to the facility.

Ongoing source identification and resolution efforts that extended into 2005 included a grease-trap inspection program that began in 2002. Originally, 68 businesses subject to grease-trap requirements were identified within the City of Morro Bay. However, in 2005, only about 50 businesses were subject to these requirements. Spot checks were conducted at over twenty of these establishments throughout the year, including the Dockside Café, Bayside Café, Rose's Restaurant, and the Inn at Morro Bay.

In May 2004, a routine grease-trap inspection performed at the Bayside Café at Morro Bay State Park resulted in the discovery of a grease buildup issue within the lift station located next to the café. Plant staff met with state park officials several times during 2004 and performed regular inspections of the grease trap to monitor it and make recommendations for improvement. During a subsequent inspection of the café's waste stream, an exceptionally elevated oil and grease reading resulted in further dialogue and meetings with state park officials to address the problem. In 2005, state park officials required the Café to install a grease interceptor, which resulted in a decrease in the amount of oil and grease entering the treatment system.

2.2 WASTEWATER CHARACTERIZATION

Monthly wastewater characterizations quantify a number of different aspects of the treatment plant's performance in 2005 (Table 2.3). Removal rates quantify the plant's performance in terms of its ability to reduce major contaminants within the wastestream. Effluent concentrations characterize the overall quality of effluent discharged through the ocean outfall, while mass emissions quantify the cumulative load of wastewater constituents that are introduced into the marine environment.

Monthly averages of the principal influent and effluent characteristics were computed from numerous samples that were collected and analyzed by treatment-plant personnel throughout 2005. The WWTP personnel regularly determine the principal physicochemical properties of the effluent, including suspended solids, BOD, pH, O&G, total residual chlorine (TRC), turbidity, settleable solids, and total coliform bacteria. The frequencies at which individual samples were collected, and the duration over which the samples were collected, varied among the parameters. For example, average reductions in suspended solids and BOD were determined from 24-hr composite samples of influent and effluent that were collected and analyzed at least weekly.

Table 2.3 Monthly Averages of Influent and Effluent Parameters

Month	Flow (MGD)	Suspended Solids Influent (mg/L)	Suspended Solids Effluent (mg/L)	Suspended Solids Removal (percent)	Suspended Solids Mass Emission (kg/day)	Biochemical Oxygen Demand Influent (mg/L)	Biochemical Oxygen Demand Effluent (mg/L)	Biochemical Oxygen Demand Removal (percent)	Biochemical Oxygen Demand Mass Emission (kg/day)
January	1.533	302	30	86.7	169	238	53	76.6	337
February	1.469	430	25	94.2	144	321	45	85.6	254
March	1.425	377	24	93.8	134	278	38	86.1	187
April	1.230	400	23	93.7	107	326	49	84.8	240
May	1.252	364	28	93.1	133	327	55	82.4	248
June	1.212	380	25	94.1	113	311	61	79.0	297
July	1.347	406	34	91.9	170	365	65	81.9	321
August	1.248	355	28	92.7	133	303	60	80.0	279
September	1.135	414	17	96.0	73	325	39	87.9	181
October	1.058	375	15	95.9	59	314	39	87.1	158
November	1.054	347	20	93.9	79	254	46	82.3	196
December	1.089	326	21	93.8	86	279	49	82.0	195
Average	**1.254**	**373**	**24**	**93.3**	**116**	**303**	**50**	**83.0**	**241**
Monthly Limitation	**≤2.36**[1]		**≤70**	**≥75.0**	**≤546**		**≤120**	**≥30.0**	**≤936**
Annual Total (MT)					**42**				**88**
Nominal Annual (MT/year)					**≤199**				**≤342**

Table 2.3 (continued) Monthly Averages of Influent and Effluent Parameters

Month	PH Influent	PH Effluent	Turbidity (NTU)	Settle-able Sol-ids (ml/L)	30-Day Median[2] Total Coliform (MPN/100ml)	Oil and Grease Influent (mg/L)	Oil and Grease Effluent (mg/L)	Oil and Grease Mass Emissions (kg/day)
January	7.6	7.6	22	<0.1	8	110.3	5.1	31
February	7.6	7.6	23	<0.1	9	127.2	6.3	39
March	7.7	7.6	20	<0.1	3	65.0	4.8	37
April	7.7	7.6	20	<0.1	2	41.0	4.1	17
May	7.6	7.6	23	<0.1	2	41.3	4.3	19
June	7.6	7.6	24	<0.1	<2	82.5	3.1	14
July	7.6	7.5	28	<0.1	2	60.8	5.6	28
August	7.7	7.5	24	<0.1	5	28.3	5.0	23
September	7.7	7.6	22	<0.1	11	57.0	3.5	14
October	7.7	7.5	21	<0.1	2	38.6	2.7	10
November	7.8	7.7	23	<0.1	3	38.5	4.3	16
December	7.8	7.6	26	<0.1	4	50.3	3.5	14
Monthly Average	**7.7**	**7.6**	**23**	**<0.1**		**61.7**	**4.4**	**22**
Monthly Limitation		**6-9**	**≤75**	**≤1.0**	**≤23**		**≤25.0**	**≤195**

[1] Peak Seasonal Dry-Weather Flow (PSDWF)
[2] Computed from samples collected in the 30 days prior to the last day of the month (MBCSD 1997a)

Analyses for the concentration of the remaining constituents were conducted only on effluent samples. Discrete effluent grab samples were analyzed for pH, coliform, TRC, temperature, turbidity, and settleable solids. Effluent grab samples were analyzed for total coliform five times each week, whereas O&G concentrations were determined weekly. Effluent ammonia was determined from monthly grab samples. Most other parameters were measured on a daily basis.

Detailed analysis of the long history of these measurements has shown that the plant has consistently outperformed expectations for wastewater treatment, based on regulatory standards. Over the two decades of monitoring, there has been no indication of deterioration in plant performance, and effluent quality has consistently exceeded expectations, based on the original design criteria. On rare occasions when exceptions to standards or criteria have occurred, they have been the direct result of a brief, unavoidable mechanical failure of a treatment-system component or the external introduction of a contaminant within the influent stream. In spite of a diligent program of preventative maintenance, the challenge for plant personnel has been to quickly respond to unanticipated failures in system components.

Because of their potential impact on effluent quality, rain events, population fluctuations, and equipment-related incidents that were experienced during 2005 are briefly described in Table 2.4. Unusual measurements in plant parameters were directly related to one of these incidents. This direct relationship between external events and effluent parameters demonstrates the overall effectiveness of the treatment process and the high degree of control that is exercised by the plant personnel under normal operating conditions. If this were not the case, excursions in effluent properties would be random and uncorrelated with external events. During 2005, none of these events resulted in an exceedance of the discharge limits specified in the NPDES permit. In the following discussions of the various wastewater characteristics, reference is made to the eleven events listed in the Table 2.4.

Table 2.4 Events that Affected the Treatment Process during 2005

#	Date(s)	Event Description	Effect on Treatment Process
E1	12/27/05-01/03/06	• Rainfall (3.1 inches) from a 8-day storm • Inundation of the treatment plant by Morro Creek on 12/31/05 • Temporary increase in population during the New Year's holiday	a) 1.5 MG of Creek water inflow into the treatment process b) 3.3 MG of rainwater inflow into the collection system c) 0.4 MG of additional wastewater throughput d) 0.5 MG of excess groundwater into collection system e) 3rd highest daily flow (2.7 MGD) f) Highest effluent TSS (65 mg/L) g) Highest TSS emission (491 kg/day) h) Lowest TSS removal (78%) i) Highest effluent BOD (81 mg/L) j) Highest BOD emission (612 kg/day) k) Lowest BOD removal (61%) l) Highest effluent turbidity (89 NTU) m) Lowest effluent pH (7.3)
E2	1/7-1/12	• Rainfall (1.2 inches) from a 6-day storm	a) 1.8 MG of rainwater inflow into the collection system b) 2.2 MG of excess groundwater into collection system c) 4th, 5th, & 6th highest flows (2.4-2.6 MGD) d) 2nd highest TSS emission (409 kg/day) e) 2nd lowest TSS removal rate (83%) f) Lowest influent BOD (170 mg/L)

#	Date(s)	Event Description	Effect on Treatment Process
E3	2/12-2/21	• Rainfall (1.9 inches) from a 10-day storm • Temporary increase in population during the President's Day holiday	a) 2.1 MG of rainwater inflow into the collection system b) 0.4 MG of additional wastewater throughput c) 3.3 MG of excess groundwater into collection system d) 2nd highest daily flow (2.9 MGD) e) 5th highest TSS emission (379 kg/day) f) 2nd highest effluent turbidity (75 NTU) g) Highest influent O&G (127 mg/L) h) Highest effluent O&G (12.4 mg/L) i) 2nd highest O&G emission (96 kg/day)
E4	3/17-3/28	• Rainfall (1 inch) from a 6-day storm • Temporary increase in tourist population during Easter spring break	a) 1.0 MG of rainwater inflow into the collection system b) 2.8 MG of excess groundwater into collection system c) 2.0 MG of additional wastewater throughput d) Highest daily flow (3.5 MGD) e) 3rd highest TSS emission (394 kg/day) f) 3rd highest influent O&G (8.6 mg/L) g) Highest O&G emission (113 kg/day)
E5	5/5-5/8	• Rainfall (0.9 inches) from a 4-day storm	a) 0.9 MG of rainwater inflow into the collection system b) 3rd highest effluent turbidity (67 NTU) c) 2nd highest residual chlorine (0.84 mg/L)
E6	5/28-5/30	• Temporary increase in the tourist population during Memorial Day weekend	a) 0.8 MG of additional wastewater throughput
E7	7/2-7/7	• Large temporary increase in the tourist population during the week of the Independence Day holiday	a) 1.6 MG of additional wastewater throughput b) 2nd highest effluent TSS (56 mg/L) c) 4th highest TSS emission (388 kg/day) d) Highest influent BOD (460 mg/L) e) 2nd highest effluent BOD (79 mg/L) f) 2nd highest BOD emission (394 kg/day)
E8	9/3-9/5	• Temporary increase in the tourist population during Labor Day weekend	a) 0.8 MG of additional wastewater throughput b) 2nd highest influent BOD (396 mg/L) c) Highest residual chlorine (0.85 mg/L)
E9	9/14-9/20	• Modification of the chlorine supply system	a) 3rd Highest residual chlorine (0.76 mg/L)
E10	11/24-11/28	• Temporary increase in the tourist population during the Thanksgiving holiday week	a) 0.5 MG of additional wastewater throughput b) 4th highest effluent BOD (73 mg/L)
E11	12/25-12/31	• Rainfall (1.3 inches) from a 7-day storm	a) 1 MG of rainwater inflow into the collection system

2.2.1 Flow Rate

Flow data collected in 2005 demonstrate that plant throughput was far below design capacity, as reflected in the discharge limitations specified in the NPDES permit. The waste discharge requirements (RWQCB-USEPA 1998a) specify that '…*the peak seasonal dry weather flow* (PSDWF) *shall not exceed a monthly average of 2.36 MGD*….' This permit limitation was not exceeded during the 2005 dry season (April through September), or in any month during 2005 (Table 2.3). The highest monthly average flow of 1.533 MGD occurred in January because of an abnormally intense rainfall event that began in late 2004 (Event E1 in Table 2.4 and Figure 2.5) The maximum daily flow during the dry season of 1.828 MGD on Independence Day (Event E7) was also well below the dry-weather monthly limit of 2.36 MGD. Accordingly, the 30-day running average (shown by solid thin line in Figure 2.5) remained below 1.44 MGD during the

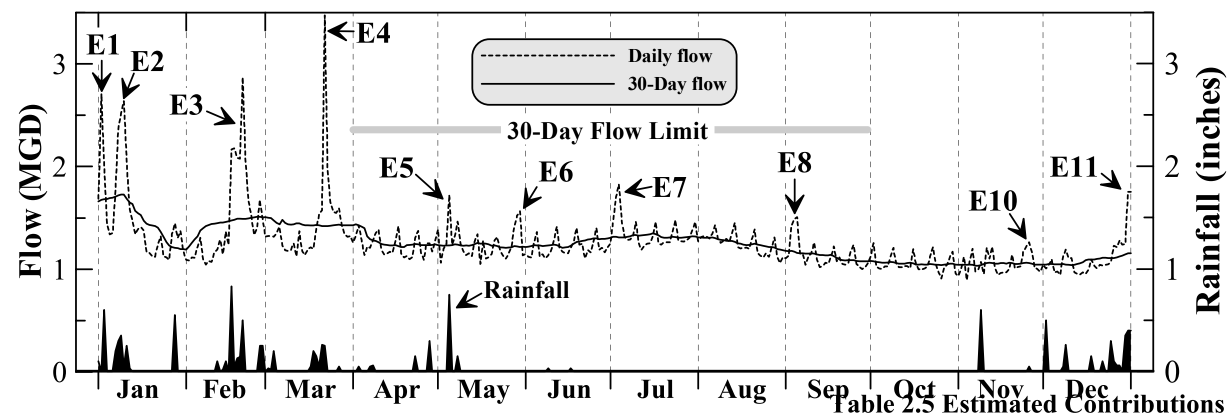

Figure 2.5 Daily Effluent Flux and Rainfall Accumulation

dry season, a level that is only 61% of the monthly permit limit (shown by the thick shaded line that spans the '*dry*' season).

Identifiable contributions to the annual average flow rate of 1.254 MGD are listed in Table 2.5. These contributions are described in the subsections below. The largest single identifiable component of flow arose from the infiltration of groundwater into the collection system. It contributed at least 0.205 MGD to the average flow through the treatment plant. Combined population increases by residents on weekends, and tourists during the summer and various holidays, accounted for approximately 10.1% of the reported annual throughput. Inflow of rain into the collection system, although significant during rainstorms, increased the flow by only approximately 2.3% during 2005.

Table 2.5 Estimated Contributions to Annual Flow Rates (MGD)

Source	Rate	Fraction
Total [1]	1.254	100.0%
Base	0.895	71.4%
Infiltration	0.205	16.3%
Weekends [2]	0.082	6.5%
Inflow	0.028	2.3%
Seasonal Tourism [3]	0.023	1.8%
Holiday Tourism [4]	0.021	1.7%

[1] From Table 2.3
[2] Increased resident population on weekends and some local tourism
[3] Increased population during the summer and early fall
[4] Population increases during holidays

The remaining 0.895 MGD of base flow was equal to the lowest reported daily flow, which was recorded on November 4th, after the summer tourist season had ended, and before the major winter storms began. However, it is likely that this entire base flow does not represent wastewater purely of sewage origin that was generated by the full-time resident populations of Cayucos and Morro Bay. Instead, because it is based on the lowest reported flow, it undoubtedly includes contributions from ongoing infiltration that occurs throughout the year, including the day with the lowest flow of the year. In fact, a much lower flow of 0.741 MGD was recorded in 2002. It suggests that at least another 0.154 MGD of "*Base*" 2005 flow can be ascribed to infiltration, bringing the infiltration contribution closer to a level exceeding 28%. The actual amount of infiltration is probably even higher than this, but it cannot be determined from the time series of daily flow rates alone. In any regard, the 2005 infiltration contribution listed in Table 2.5 probably significantly underestimates the true level of infiltration.

Overtotalized Flow

In 2005, overtotalization of daily flow volumes was negligible. The general absence of overtotalized flow contrasts markedly with the artificially elevated daily flow rates reported prior to 2002 (shaded portion of Figure 2.6). Before July 2001, flow rates were reported using the effluent meter. Because it routinely overestimated discharge rates by at least 25%, an influent flow meter was installed and tested in 2000. Although it was consistently more accurate than the effluent flow meter, it, too, would overtotalize the flow on rare occasions when hydraulics in the influent flume were compromised. Influent flow was measured using a prefabricated metering manhole and an ultrasonic transducer capable of highly accurate flow determinations based on precise water-level measurements. However, when the water level within the headworks wetwell exceeded a certain point, influent would backup along the 350 ft of pipe between the treatment plant and the metering flume. The resulting increase in water level within the flume was misinterpreted as increased flow by the ultrasonic depth transducer.

Based on recommendations resulting from analyses performed on 2002 flow data (MRS 2003b), and beginning in 2003, an adjusted effluent flow measurement was reported instead of the flow measured by the influent meter on rare occasions when the metering flume was surcharged. Not surprisingly, in 2003, 2004, and 2005 there were only a few occasions when adjusted effluent measurements were reported in

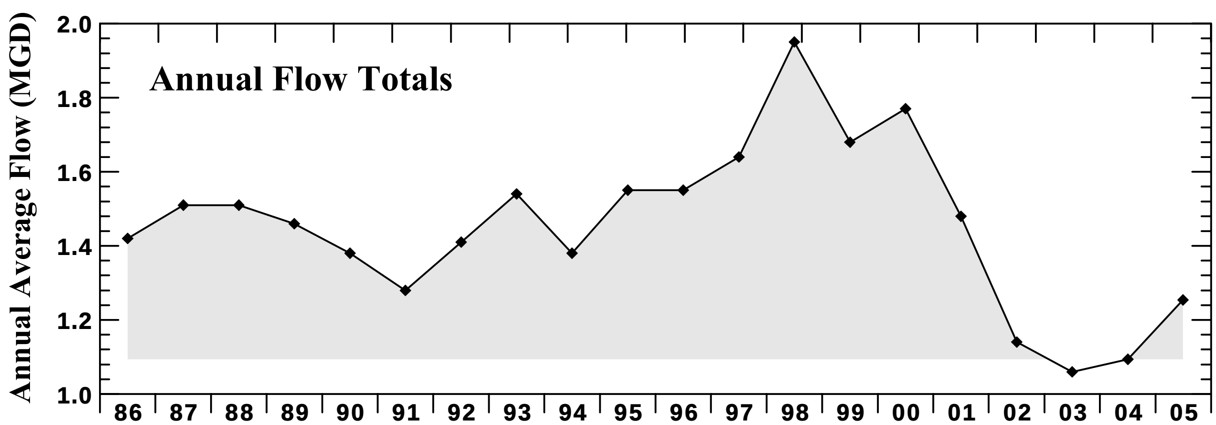

Figure 2.6 Annual Average Flow Rate

place of erroneous influent measurements.

Inflow and Infiltration

Throughout 2005, a measurable portion of influent came from inflow (direct runoff of rainwater into the collection system) and infiltration (seepage of groundwater into the collection system). Processing this relatively clean inflow and infiltration (I&I) does not tax the treatment capabilities of the plant to the same extent as does processing equivalent volumes of pure sewage. Municipal sewage is higher in total suspended solids (TSS), biochemical oxygen demand (BOD), and other organic contaminants customarily associated with wastewater. This is why NPDES maximum flow limits are typically applied to measurements collected during the '*dry season*,' when contributions from I&I are expected to be negligible. It is thought that this is when the treatment process is subjected to the greatest concentrations of organic particulates.

However, the flow recorded at the MBCSD WWTP throughout 2005 included measurable I&I and was not representative of a wastestream (that is, pure sewage) that challenges the treatment process. By ignoring the I&I component, one would incorrectly conclude that the plant had reached approximately 60% of its design capacity, namely, the measured flow averaged over the '*dry*' season (1.24 MGD) relative to the design capacity (2.06 MGD). However, at least 0.12 MGD, and probably much more of the measured average dry-weather flow arose from infiltration, so the plant was probably well below 55% of its capacity to process wastewater entirely of sewage origin.

Rainfall-induced increases in daily plant flow appear as a series of isolated spikes in the dashed line of Figure 2.5 on Page 2-24, particularly during the first three months of 2005. Their timing relative to major rainfall events (shown in solid black at the bottom of the Figure) demonstrates that they were caused by surface runoff flowing into the collection system. Overall, the 10.62 inches of rain that fell during 2005 generated an additional 10.4 million gallons of flow to the collection system. This indicates that approximately one million gallons of additional flow through the treatment plant were generated by each inch of rain that fell. This rate of inflow would be produced by runoff entering the collection system from a 36-acre drainage area. Although this is a small fraction of the area covered by the collection system, it was twice the drainage area from inflow recorded in 2003, but comparable to the inflow rate measured in 2004.

It is also noteworthy that a number of inflow events were associated with rainstorms that occurred during the '*dry season*.' This is evident from the rainfall and associated elevated flow measurements that occurred throughout April and early May as shown in Figure 2.5. One particularly significant rain-fall events (E5) that occurred during the dry season is easily discerned as a brief but perceptible increase of nearly 1 MG in plant flow during early May. Other rainfall-induced episodic increases in plant flow appear as a series of spikes in the dashed line of Figure 2.5, particularly at the beginning and end of 2005.

In addition to episodic inflow of rainwater into the collection system, rains also increase infiltration of groundwater into the system. In contrast to inflow events, infiltration extends beyond the duration of the rainfall event. Winter rainfall recharges the aquifers and raises the water table above the shallower sections of the collection system. As a result, groundwater infiltration increases. As the rains dissipate through the dry season, the contribution of groundwater infiltration declines as the water table drops, but the contribution of sewage-derived wastewater increases due to seasonal increases in tourist populations. As shown by the solid black line in Figure 2.5, these offsetting contributions to plant flow maintain a relatively constant 30-day running-mean flow between mid-April and July. In August and September, declining contributions from both the tourist population and infiltration resulted in a steady decline in average flow. These declines ultimately led to a minimum in 2005 flow rates during November.

The long-term decrease in flow averages during 2005 indicates that groundwater infiltration contributed at least 0.18 MGD (14.3%) to the plant throughput. However, in addition to the long-term contributions from groundwater infiltration, an additional component of infiltration arose from the intense rainstorms in late 2004 and early 2005. Periods of intense rainfall between December 2004 and March 2005 caused the water table to temporarily increase to an unusually high level, and even extend above ground level in certain locations. The impact of this additional infiltration is evident in the difference in the extended duration (width) of the flow events (dashed line) shown in Figure 2.5, as compared to the associated rain events (solid black at the bottom of the figure). This infiltration contributed an additional 2% to the plant's average annual throughput, bring the total infiltration contribution to 0.2 MGD (16.3%). However, as described above, infiltration undoubtedly accounts for some of the plant flow that was measured below minimum measured flow in 2005 (0.895 MGD). Consequently, infiltration undoubtedly accounts for much more than the 0.205 MGD shown in Table 2.5.

Population-Related Flow Events

In 2005, three rain events coincided with increases in tourist populations (Events E1, E3, and E4 in Table 2.4 and Figure 2.5). As a result, precise proportions of short-term flow increases that were solely due to population fluctuations were more difficult to determine in these combined events. Nevertheless, historical flow increases from increased tourist populations during the same holidays provide a sound basis for estimating population-related short-term increases in plant flow.

The economy of MBCSD is heavily influenced by tourism, as is the flow of sewage-derived wastewater through the treatment plant. The presence of over 50 restaurants in the small community of Morro Bay bears testimony to the importance of tourism. Its influence is apparent in the population fluctuations of Morro Bay and Cayucos that increase significantly on holidays and weekends. These population fluctuations are reflected in the wastewater output from the treatment plant. The increases in flow resulting from weekend visitors and permanent residents that remain home on the weekends are evident in the daily flow measurements shown by the dashed line of Figure 2.5. They appear as regular weekly oscillations with amplitudes of approximately 0.076 MGD that occur throughout the flow record. Averaged over the year, these weekend population increases added approximately 0.082 MGD (6.5%) to the annual average flow rate.

Superimposed on these weekly oscillations are larger, episodic increases that coincide with major holidays (Table 2.4). Of particular note were the population-related Events E4 (Spring Break), E6 (Memorial Day), E7 (Independence Day), E8 (Labor Day), and E10 (Thanksgiving). On average, these population increases added approximately 0.18 MGD to the flow rate over the span of the holiday. However, when averaged over the entire year, holiday tourism added only approximately 0.021 MGD, or 1.7%, to the annual average flow.

Nevertheless, some of these short-duration population increases stand out in the daily flow totals shown in Figure 2.5. For example, increased tourist populations on the three-day Memorial Day weekend at the end of May (Event E6) added approximately 0.8 million gallons of sewage to the collection system. A similar volume was generated by the population increase over Labor Day (Event E8). In contrast, the Fourth-of-July holiday (Event E7) flow increase was particularly significant because it contributed an additional 1.6 MG to the WWTP flow. In Cayucos, where a parade, barbecue, and fireworks display draw large crowds during the Independence-Day holiday, the population increased by approximately 10,000 people. This is more than three times the estimated population of permanent residents.

Over longer periods, the influence of increased tourist and resident populations in the summer is reflected by the perceptible increase in the 30-day average flow rate shown as the solid line in Figure 2.5. Beginning in mid-June, long-term flow averages began to steadily increase relative to baseline levels. This trend continued into early August, when long-term flow rates reached a local maximum of 1.34 MGD. Long-term flow rates steadily declined through the end of September. Integrated over this time frame, the total contribution of this seasonal tourist population to the annual average flow was approximately 0.023 MGD (Table 2.4). Seasonal tourism generated approximately 1.8% of the influent processed by the treatment plant in 2005.

2.2.2 Suspended Solids, Turbidity, and Settleable Solids

Suspended solids, turbidity, and settleable solids are measures of the particulate load within the wastewater stream. One of the primary functions of the treatment process is to remove organic particulates from the wastestream. The treatment plant's high overall performance during 2005 is reflected by its removal of over 93% of the total suspended solids (TSS) from influent (Table 2.3). This is far better than is required by the NPDES permit, which necessitates removal of only 75% of the suspended solids on a monthly basis.

Accordingly, the average annual TSS concentration of only 24 mg/L was well below the monthly permit limit of 70 mg/L. Benchmark monthly removal rates in all months exceeded 85%, which is the limit established for full secondary treatment (Table 2.3 on Page 2-21). Figure 2.7d shows that the running 30-day average TSS concentration was also at or below the full-secondary standard during a great majority of the year. Overall, the monthly TSS data demonstrate that the treatment plant routinely performs at or above secondary treatment levels for both TSS removal rates, and TSS effluent concentrations.

The light sediment loads in discharged wastewater are reflected in the diffuse particulate cloud that is apparent in photograph of the diffuser shown in Figure 2.2 on Page 2-6. Over the entire year, only approximately 42 metric tons of suspended solids were discharged to the ocean. This mass emission represented 21% of the 199 metric tons that would be discharged with a permit-specified TSS concentration of 70 mg/L and the allowed average dry-weather flow rate of 2.06 MGD.

High removal rates are difficult to achieve when TSS loads within influent are already low. Regulators recognize that high removal rates are unnecessary when effluent has already achieved low TSS concentrations. Consequently, the NPDES discharge permit imposes the 75% monthly removal rate only when effluent TSS concentrations exceed 60 mg/L (RWQCB-USEPA 1998a, SWRCB 1997). Monthly effluent TSS levels for the MBCSD WWTP were well below 60 mg/L throughout 2005 (Table 2.3, Figure 2.7d). Thus, although the plant was not subject to the 75% monthly removal rate during any month of 2005, it consistently met this removal criterion.

At least in part, low influent TSS results from rainwater inflow and groundwater infiltration. Rainwater is relatively devoid of organic particulates, yet the plant achieved high TSS removal rates even during major rainstorms. The lowest removal rate (78.11%), which was measured on 3 January (dashed line in Figure 2.7b) exceeded the 75% minimum removal benchmark, and was far above the 30% minimum required for primary treatment (USGPO 1997a). This particular removal-rate measurement was partly a consequence of a comparatively low influent TSS concentration (303 mg/L in Figure 2.7c) associated with flow Event E1. This event was caused by an extended period of heavy rain, which resulted in high inflow and flooding in the treatment process.

Two other event-related factors may have also contributed to the low removal rate, namely, a reduction in removal efficiency due to the inundation of the process stream, and the increased presence of terrigenous rather than organic particulates. The treatment process is designed for the removal of sewage particulates rather than the removal of naturally occurring suspended sediments that are entrained in large volumes of rainwater runoff. In any regard, this event demonstrates how measured removal rates can be influenced by changes in the influent TSS concentration, the composition of the suspended particulates (terrigenous versus sewage), and the flow rate. Under normal operating conditions, the treatment process is able to consistently extract a much higher proportion of the suspended solid load as reflected by the level of the dashed line in Figure 2.7b between mid-January and the end of the year. Note that the axis is inverted so lower levels of the plotted line represent higher removal rates.

Over shorter periods, inflow events are readily apparent in the record of daily TSS emissions (Figure 2.7a). Events E1 through E5 were all major rainfall events that resulted in marked temporary increases in plant flow (Figure 2.5, Table 2.4). Except for Event E1, which was described in the previous paragraph, the other four episodic increases in TSS mass emissions resulted largely from an increase in flow rate, rather than an increase in effluent TSS concentrations. This is apparent from a comparison of the records of the emissions, where the events are readily apparent (Figure 2.7a), and the effluent concentrations, where the events are not as apparent (Figure 2.7e). The absence of reductions in removal rates during these events (Figure 2.7b) also indicates that the increase in emissions was due to a higher flow volume rather than any degradation in plant performance.

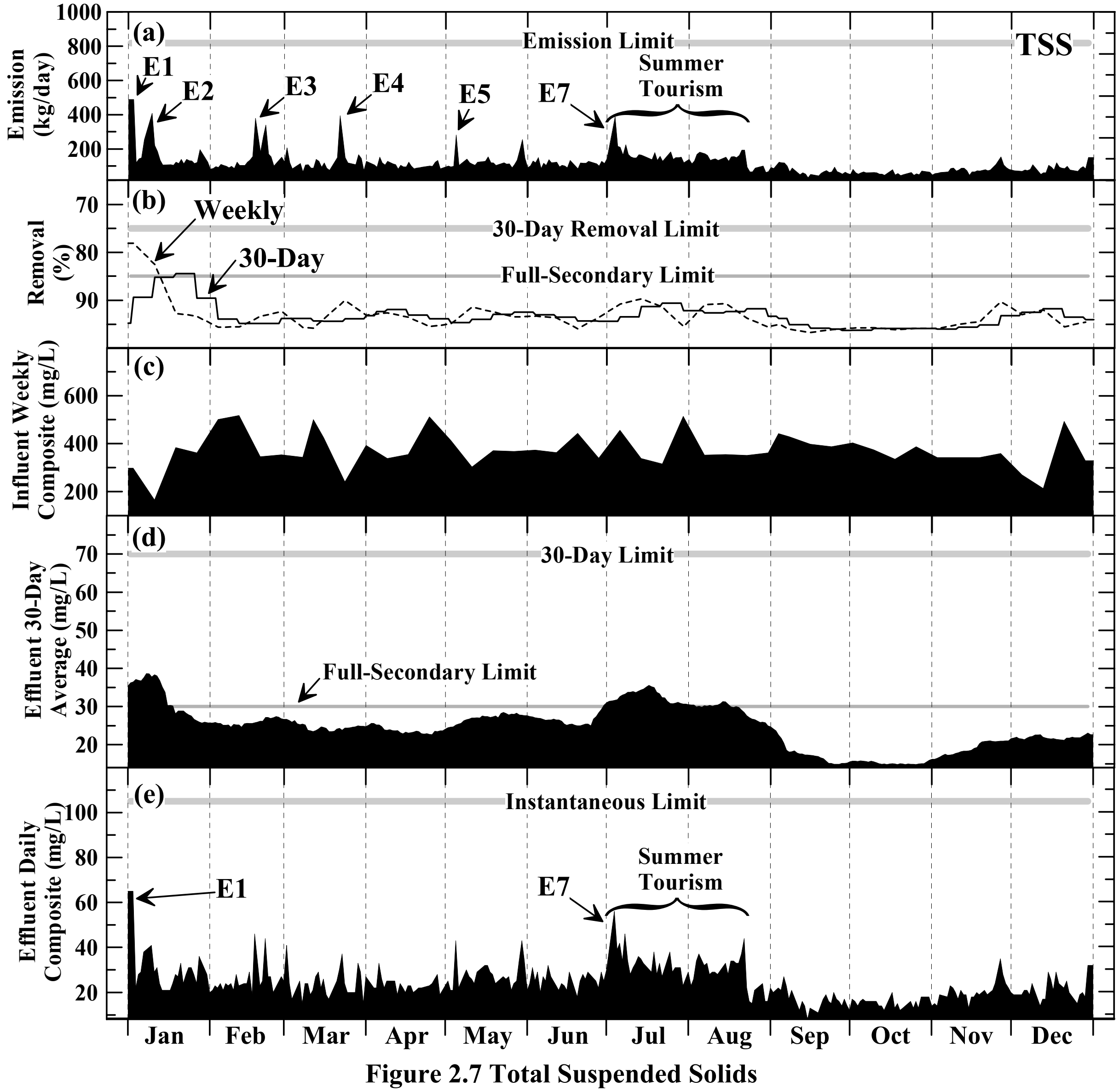

Figure 2.7 Total Suspended Solids

In contrast to rainwater inflow events, increased TSS emissions associated with population events were associated with increased TSS concentrations within both influent and effluent. In particular, the population increase during the Independence Day holiday (Event E7) coincided with increased TSS concentrations within effluent (Figure 2.7e) and increased emissions (Figure 2.7a). In contrast to inflow events, the additional TSS loading during the population events arose from organic particulates within sewage-derived wastewater rather than natural sediments suspended within rainwater runoff. Higher-than-normal flow rates also coincided with tourism events (Events E6, E7, E8, and E10 in Figure 2.5 on Page 2-24). Nevertheless, removal rates remained comparatively uniform (Figure 2.7b), indicating that the treatment plant continued to perform at high levels despite the increased organic loads.

Over long periods, the influence of increased organic loading from the summer tourist populations is apparent in effluent TSS concentrations (Figure 2.7e) and emissions (Figure 2.7a). It is particularly apparent in the 30-day running mean concentration (Figure 2.7d) during the months of July and August. Again, however, this increase in long-term loading had little effect on the plant's ability to remove solids from the process stream (Figure 2.7b)

The effectiveness of the treatment plant's screening, grit removal, sedimentation, filtration, and clarifying processes was also reflected by the record of consistently low effluent turbidity shown in Figure 2.8, and in the general absence of detectable settleable solids within the effluent (Table 2.3). In fact, none of the 365 effluent samples contained a detectable amount of settleable solids. Figure 2.8 shows that the individual turbidity measurements, as well as their averages, were all well below their respective permit limits. There were a few isolated instances when turbidity measurements were above the norm (Figure 2.8c). Six events (E1 through E5, and E11) were directly associated with rainwater inflow events, while the remaining anomalous turbidity measurement (E7) was related to the population increase on Independence Day.

The highest recorded turbidity (89 NTU on 1/3/05) occurred in conjunction with the major rainstorm (Event E1) that caused Morro Creek to overflow its banks and flood the treatment plant. The increased turbidity measured at that time was directly attributable to the presence of large amounts of turbid creek water within the plant process. Plant operators noted that the plant process had an unusual light-brown, muddy appearance, identical to that of the rain-swollen waters of Morro Creek. In fact, streams and near-shore coastal waters throughout the region had a similar muddy appearance as a result of the heavy particulate loads that were washed down waterways during the intense rainstorm. Despite the unusual nature of this turbidity measurement compared to conditions in 2005, it was still less than 40% of the instantaneous permitted limit of 225 NTU. Additionally, because this and the other turbidity events were of comparatively short duration, the seven-day and 30-day average concentrations remained well below their respective permit limits.

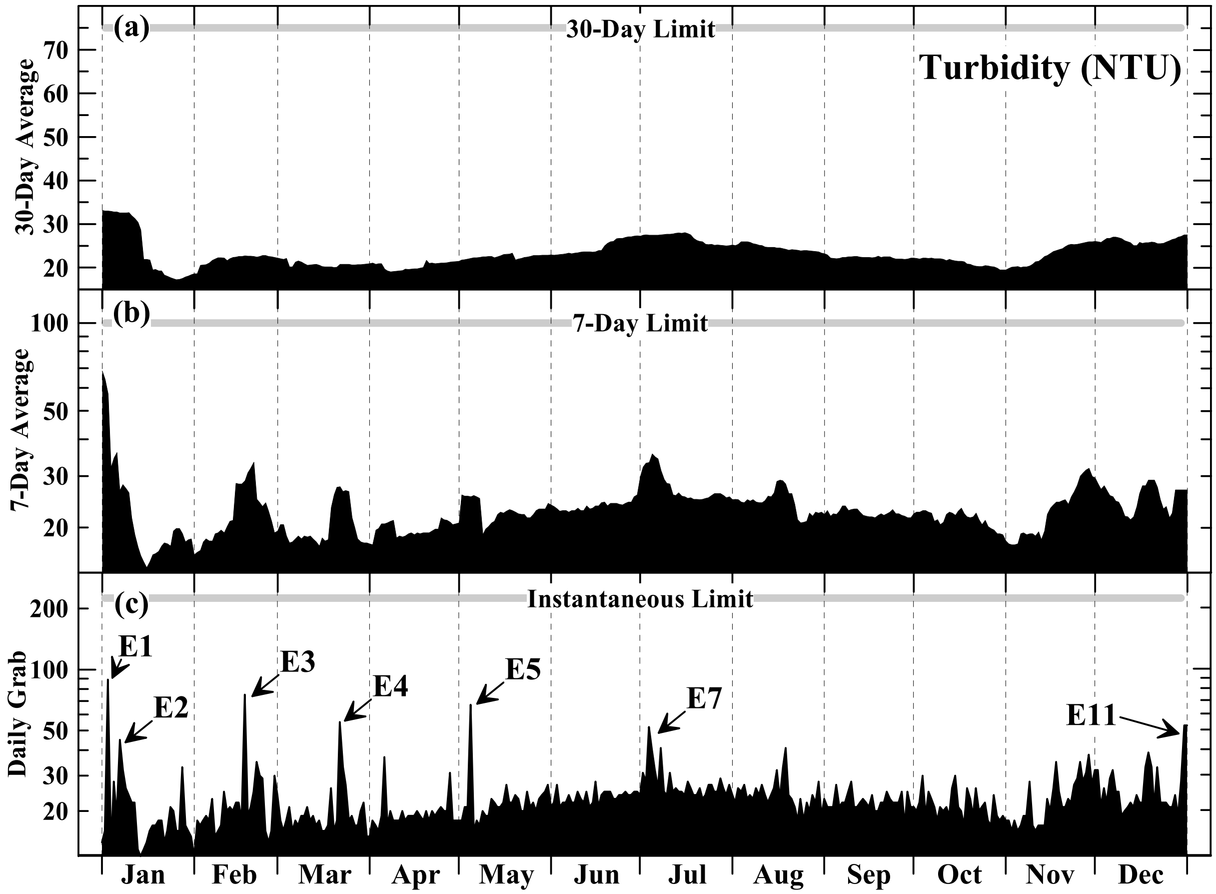

Figure 2.8 Effluent Turbidity

2.2.3 Biochemical Oxygen Demand

Biochemical oxygen demand (BOD) measures the level of organic loading within the wastewater stream. Organic loading is indirectly determined from the amount of oxygen required for aerobic bacteria to decompose organic matter in a sample of wastewater. Organic material, which supports bacterial degradation and demands oxygen, can lead to harmful environmental effects if the process of organic decomposition severly depleats the dissolved oxygen within receiving waters.

During 2005, the treatment process reduced influent organics by an average of 83%, as determined from the weekly composite samples analyzed for BOD (Table 2.3 on Page 2-21). This was more than twice the minimum monthly limit in the NPDES discharge permit, which requires removal of at least 30% of the BOD material. Even the lowest instantaneous removal rate of 61% (Event E1 in Table 2.4 on Page 2-22) was more than double the 30% minimum required for primary treatment (USGPO 1997a). Because of the high removal rates, total mass emission of oxygen-demanding organic matter during 2005 (88 MT) was less than a third of the annual discharge limit of 342 MT.

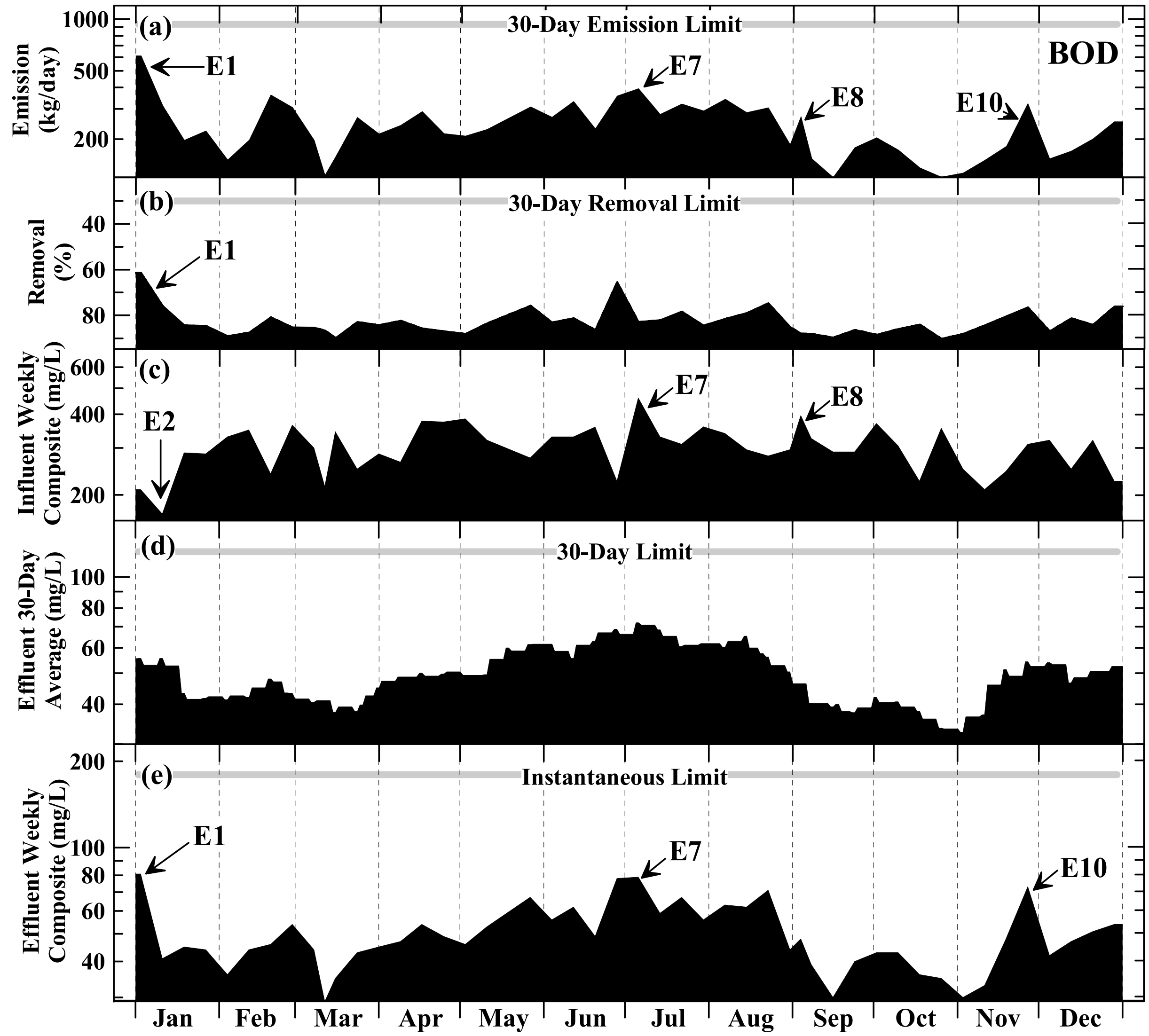

Figure 2.9 Biochemical Oxygen Demand

In four months, BOD removal rates reached or exceeded 85%, which is the requirement for full secondary treatment. In other months, BOD removal rates were close to the 85% limit established for full secondary treatment. Thus, the treatment plant routinely performs at near-secondary treatment levels insofar as its ability to remove oxygen-demanding organics from the wastestream. The treatment process also maintained average effluent BOD concentrations at or below 60 mg/L during ten months of 2005 (Table 2.3 on Page 2-21). As they do with TSS removal, regulators recognize that high BOD removal rates are unnecessary when effluent has achieved a low BOD concentration. Consequently, the NPDES discharge permit imposes the 30% minimum monthly removal rate only when effluent BOD levels exceed 60 mg/L. Thus, although the plant was not subject to the 30% removal rate during ten months of the year, it consistently achieved far higher rates of BOD removal (>76.6%).

At no time during 2005 did measured BOD concentrations in the effluent exceed the 30-day limit of 120 mg/L, much less the permitted instantaneous maximum of 180 mg/L (Figure 2.9e). Although fluctuations in effluent BOD concentration occurred throughout the year, only three of the excursions coincided with events (E1, E7, and E10 in Figure 2.9e). The highest effluent BOD coincided with plant flooding (Event E1) that affected the treatment process and resulted in the lowest BOD removal rate (Figure 2.9b) and highest BOD emission (Figure 2.9a). On Independence Day, increased organic loading from tourist populations (Event E7), led to the highest measured influent BOD (460 mg/L in Figure 2.9c) and corresponding increases in effluent BOD concentration and mass emission. A similar increase in organic loading within the influent was associated with increased tourist populations during the Labor Day and Thanksgiving holidays (Event E8 and E10 in Figure 2.9ce), which resulted in an isolated increase in emissions (Figure 2.9a). However, during those events, BOD removal rates remained relatively constant (Figure 2.9b).

2.2.4 pH

The permit requires that the hydrogen-ion concentration (pH) of the effluent remain within a range of 6 to 9 pH units at all times. Section 301(h) of the Clean Water Act allows an NPDES discharge permit to be issued that exceeds these pH limitations. However, the discharge permit issued to the MBCSD does not allow such an exception because the plant's partial secondary treatment can routinely treat wastewater to the pH standards that apply to full secondary treatment. Moreover, the general absence of heavy industrial input into the collection system results in an influent stream with nominal pH that already meets the discharge requirements without treatment.

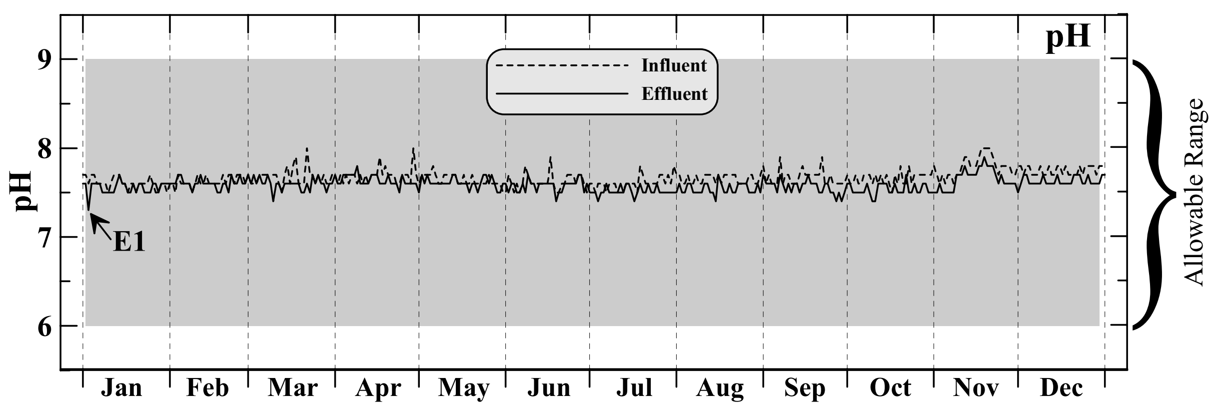

Figure 2.10 Hydrogen-Ion Concentration

As a result, effluent pH measurements remained within the allowable range throughout 2005 as shown by the solid line in Figure 2.10. The treatment process moderated the amplitude of some of the short-term fluctuations that occurred in the influent pH. As a result, the average pH of effluent was 0.082 pH units lower than influent, and the influent coefficient of variation (CV) of 1.2% was reduced to 1.1% by the treatment process. Thus, the effluent was slightly less alkaline and more stable than the influent. It was also closer to the neutral hydrogen-ion concentration of pure water.

The only unusual pH recording was a slightly reduced effluent pH (7.3) measured on January 3rd in conjunction with Event E1 (Figure 2.10). As with the other unusual readings associated with Event E1, the slightly more acidic conditions within the process stream at that time were related to the inundation by Morro Creek.

2.2.5 Oil and Grease

During 2005, the treatment process reduced average O&G concentrations within the influent stream by a factor of 10 (Table 2.3). The resulting effluent concentrations remained low, only exceeding 8.5 mg/L on three occasions over the course of the 52 weekly measurements. The samples collected on January 11th (Event E2), February 22nd (Event E3), and March 22nd (Event E4), had O&G concentrations of 10.7 mg/L, 12.4 mg/L, and 8.6 mg/L respectively. While these three measurements were substantially higher than most other measurements, all three concentrations were still more than six times lower than the instantaneous permit limit of 75 mg/L. It is not immediately evident why the effluent O&G was abnormally high in the January 11th sample (Event E2), but the increased O&G concentration in the February and March samples were probably related to the increased service-area population during the President's Day holiday (Event E3) and spring break (Event E4). In any regard, all of the measurements were well below the instantaneous limit of 75 mg/L, and despite the high flow rates caused by rainwater inflow during these events, the associated emissions of 82 kg/day (E2), 96 kg/day (E3), and 113 kg/day (E4) were only a fraction of the permitted maximum emission of 585 kg/day (Figure 2.11b).

The remaining O&G concentrations were all less than a fifth of the weekly average limitation of 40 mg/L, and far below the maximum daily limit of 75 mg/L. Similarly, the highest maximum monthly average of 6.3 mg/L, which occurred in February 2005, was less than a third of the 30-day limitation of 25 mg/L specified in the NPDES discharge permit (Table 2.3).

2.2.6 Temperature

Although there are no limits imposed on effluent temperature, it is an important physical property to

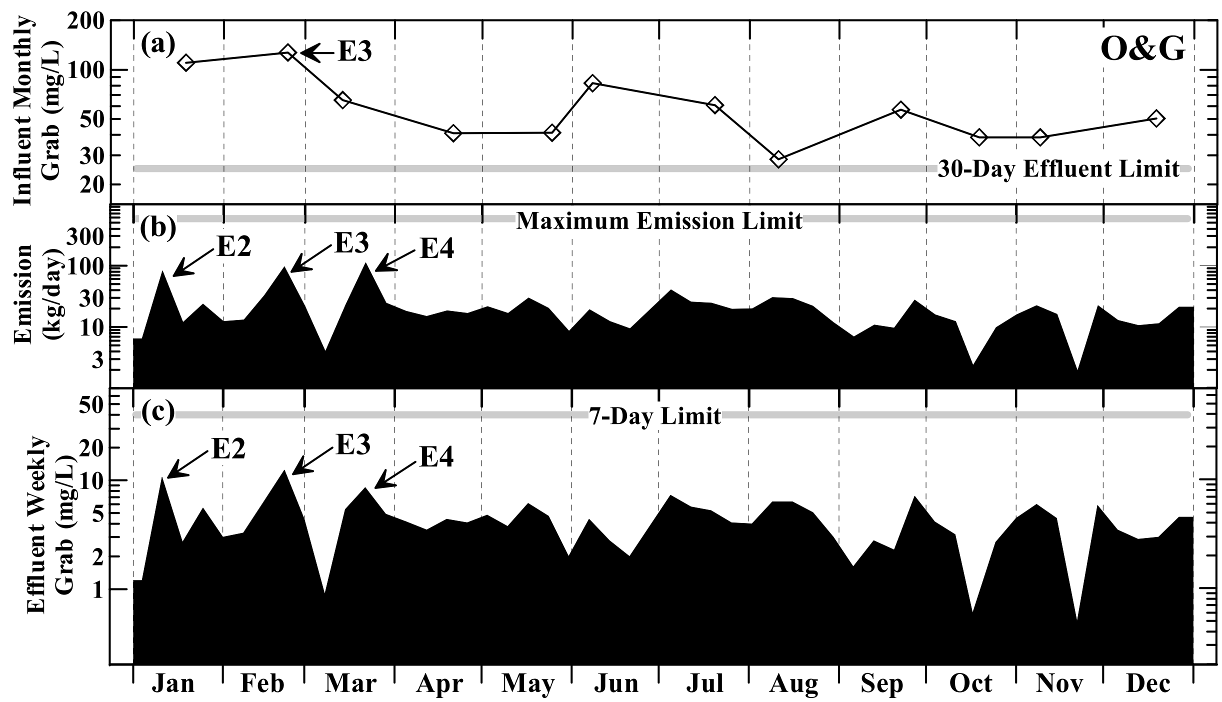

Figure 2.11 Oil and Grease

document. The difference between effluent and receiving-water temperature dictates the initial dilution of the wastewater plume within the ocean. As with receiving waters, effluent temperature exhibits a distinct semi-annual cycle with maximum temperatures between mid-July and mid-October (Figure 2.12).

Effluent temperature reached its lowest level (15ºC) on 14 January, shortly after an unusually cold winter rainstorm (Event E2) past through the area. For the most part, however, temperatures were comparatively

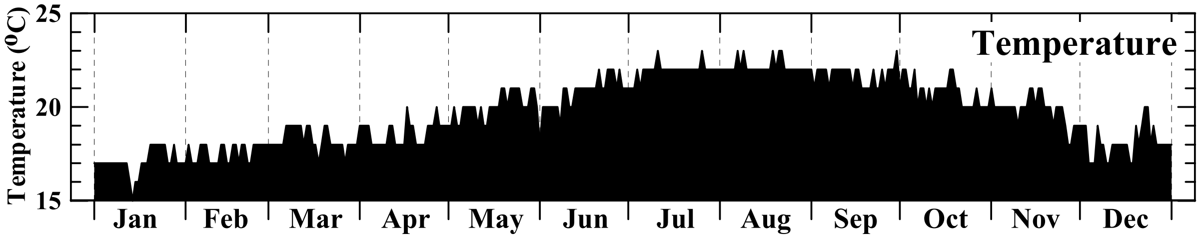

Figure 2.12 Effluent Temperature

constant, between 17 and 18ºC, through mid-April. The gradual temperature increase from spring to early summer resulted from increased ambient insolation. In the fall, effluent temperatures declined over a span of approximately two months (October and November).

2.2.7 Ammonia

The concentration of ammonia as nitrogen (NH_3-N) was uniformly low throughout 2005. Grab samples that were analyzed for ammonia on a monthly basis had concentrations that were at or below 33 mg/L (Figure 2.13). Peak concentrations were 24 times lower than the permitted instantaneous limit of 804 mg/L imposed for the protection of marine aquatic life, and less than half of the 6-month median limit of 80.4 mg/L.

2.2.8 Residual Chlorine

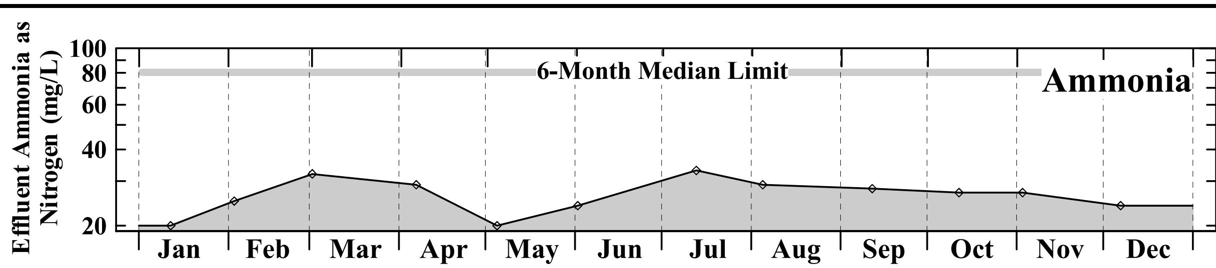

Figure 2.13 Effluent Ammonia

Total residual chlorine (TRC) is a measure of the chlorine remaining in the effluent following disinfection with sodium hypochlorite and subsequent dechlorination or buffering with sodium bisulfite. Daily grab samples for chlorine residual were collected at times when concentrations of effluent constituents are expected to be at their highest levels, namely, during peak flow. Balancing chlorination and dechlorination to obtain adequate disinfection (coliform reduction) without dosing the marine environment with high levels of chlorine can be challenging. The chlorine demand is constantly changing due to continuous variations in flow rates. Consequently, the disinfection process occasionally requires direct operator intervention. If too much sodium hypochlorite is added, beyond what is needed to meet the coliform demand, then dechlorination may be made difficult and TRC levels in the effluent may increase.

Throughout 2005, treatment plant personnel maintained careful control over the disinfection-dechlorination process. Only 5%, or 19, of the 365 daily grab samples had detectable concentrations of effluent TRC. Consequently, the running 6-month median TRC never exceeded the 0.05-mg/L detection limit. This detection limit is five-times lower than the 6-month median limit of 0.27 mg/L specified in the discharge permit. Additionally, on no occasion during 2005 did measured TRC concentrations exceed the permitted maximum daily limit of 1.07 mg/L, which was established to protect marine aquatic life (Figure 2.14).

There were three occasions when TRC concentrations exceeded 0.6 mg/L. The two highest concentrations, which were near 0.85 mg/L, coincided with external events (E5 and E8 in Table 2.4). The first

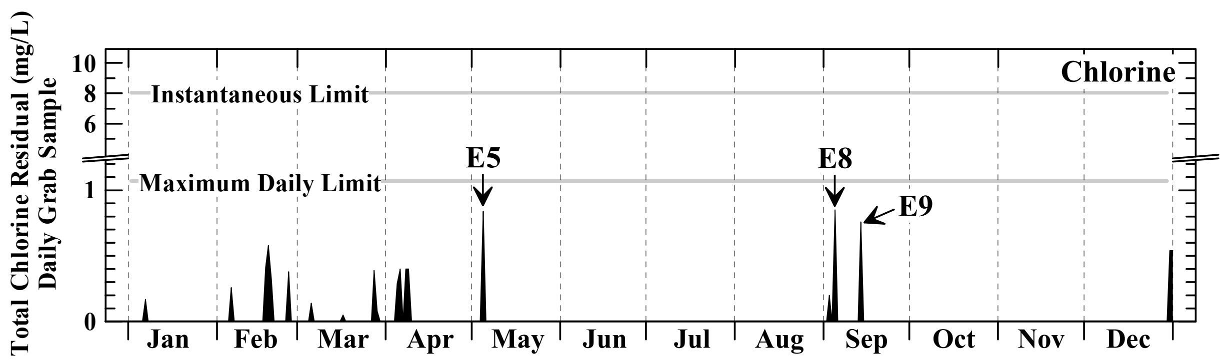

Figure 2.14 Effluent Total Residual Chlorine

event was a late season rainfall event and the second event was a temporary population increase associated with tourism surrounding the Labor Day holiday. The third event (E9) on 14 September resulted in a TRC concentration of 0.76 mg/L. This slightly elevated TRC concentration occurred during modifications to the supply piping in the disinfection system. While the system was offline, sodium hypochlorite was manually infused into the wastestream at the surface of the contact chamber. Because this chlorination method is less precise than the automated system, measured TRC concentrations deviated from the norm on that day.

2.2.9 Coliform Bacteria

Effluent samples were collected on five consecutive days each week and analyzed for the most probable number of total coliform organisms per 100 ml (MPN/100ml). The monthly median coliform densities computed at the end of each month (Table 2.3) were all less than half of the 30-day median limit of 23 MPN/100ml specified in the NPDES permit. Figure 2.15 shows that measured daily coliform densities were also well below the permitted instantaneous maximum of 2400 MPN/100ml. In fact, a third of the 284 effluent samples analyzed in 2005 had no measurable coliform. The highest coliform density of 900 MPN/100ml was measured on 28 October. This and the handful of other slightly elevated measurements of coliform density did not bear any relation to events that affected the treatment process.

2.2.10 Effluent Toxicity

Effluent samples were tested quarterly and semiannually for toxicity using two types of bioassays.

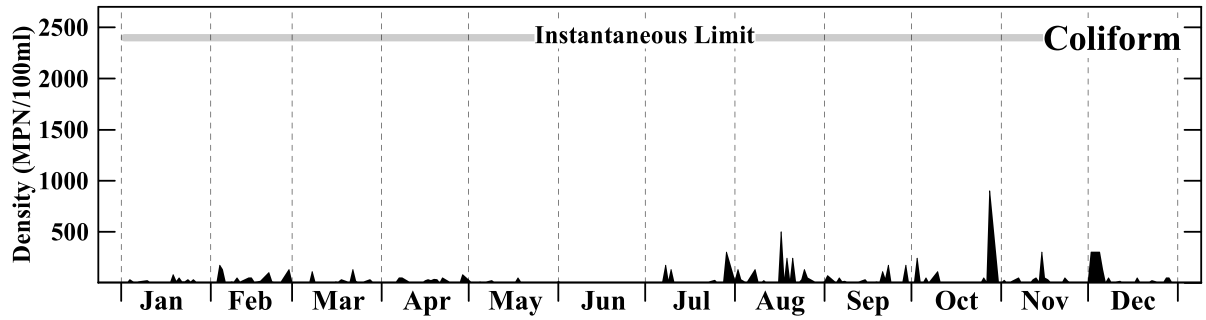

Figure 2.15. Effluent Total Coliform

Chronic bioassays were conducted on composite effluent samples collected in January and July (Table 2.6), while acute toxicity tests were conducted on effluent grab samples collected in January, April, July, and October (Table 2.7). The chronic bioassay tests found consistently low effluent toxicity, with levels far below limits specified in the discharge permit. Although the acute bioassays also measured toxicity levels that were consistently below permit limits, the tests were confounded by ammonia interference, which artificially elevated the reported results.

Table 2.6 Giant Kelp (*Macrocystis pyrifera*) Chronic Bioassay Analyses

Criteria	Duration	NOEC (%)	TUc	Permit Limit TUc
10 January 2005				
Germination	48 hour	18	5.6	134
Growth	48 hour	18	5.6	134
11 July 2005				
Germination	48 hour	18	5.6	134
Growth	48 hour	18	5.6	134

The chronic toxicity tests conducted in January and July measured spore germination and growth response in giant kelp (*Macrocystis pyrifera*) after exposure to a range of effluent dilutions. The results of bioassays conducted in 2005 are summarized in Table 2.6 and demonstrate the effluent's low chronic toxicity during 2005. Both chronic-toxicity endpoints were more than an order of magnitude lower than the applicable permit limitation of 134 TUc for the daily maximum toxicity. The reported chronic toxic units (TUc) were based on a '*No Observable Effects Concentration*' (NOEC), which is the highest effluent concentration that does not cause an adverse effect statistically different from a control sample. They indicate that the chronic bioassays did not detect adverse effects in marine organisms exposed to effluent concentrations at or below 18%. The permit allows adverse effects in concentrations as low as 0.75%.

The raw test data, pertinent QA/QC data, and chains of custody for the chronic bioassays were provided in the semiannual effluent reports (MRS 2005gh). As part of quality control, chronic bioassays were also conducted using a reference toxic chemical (copper chloride) to determine the sensitivity of these particular sets of giant kelp sporophyll (*M. pyrifera*) to toxins. QC results demonstrated that the kelp specimens were highly susceptible to toxic exposure because they experienced a significant reduction in both germination and growth in concentrations as low as 32 µg/L of the reference toxicant. The concomitant germination and growth NOEC's for the reference toxicant were 18 µg/L, which was comparable to toxic concentrations found in tests from prior years. This suggests that these particular batches of kelp spores were sensitive to toxic materials. Toxic screening studies conducted in 1993 established that giant kelp (*M. pyrifera*) is substantially more sensitive to MBCSD effluent than other species, such as the larvae of the

Table 2.7 Fathead Minnow (*Pimephales promelas*) Acute Bioassay

Date 2005	LC_{50} (%)	Toxicity Concentration (TUa)[2]	Permit Limitations (TUa)[1] 30-day Average	7-day Average	Instantaneous Maximum
10-13 January	>100	0.76	1.5	2.0	2.5
11-14 April	70.7	1.41			
11-14 July	70.7	1.41			
10-13 October	70.7	1.41			

[1] Permit limitations for acute toxicity were based on an outdated version of the COP (SWRCB 1997). The current COP (SWRCB 2001) sets the daily TUa limit at 4.29.

[2] Acute toxicity levels were artificially elevated due to unavoidable ammonia interference introduced by a change in bioassay protocol that was required by the RWQCB (2003b)

inland silverside (*Menidia beryllina*) and bay mussel (*Mytilus edulis*) (MRS 1994). Nevertheless, chronic bioassays conducted during 2005 demonstrated that giant kelp was only minimally affected by exposure to treatment plant effluent.

Although chronic bioassays are far more sensitive and accurate than acute tests, the current NPDES discharge permit requires acute bioassays in addition to the chronic tests described above. However, in lieu of chronic bioassays, the COP (SWRCB 2001) no longer requires that acute tests be conducted on effluent with a dilution ratio comparable to that of the MBCSD discharge. Moreover, when acute bioassays are conducted, the latest COP imposes limitations that are fundamentally different from those specified in the current NPDES issued to the MBCSD. For the MBCSD discharge, the new daily TUa limit of 4.29 is almost three times higher than the 30-day average limitation (TUa=1.5) specified in the current NPDES discharge permit.

One of the major drawbacks to acute bioassays conducted on wastewater samples is that they are easily confounded by presence of ammonia. Ammonia interference strongly influenced the results of the acute bioassays conducted in 2005 (Table 2.7). The acute bioassays examined the mortality of the Fathead Minnow (*Pimephales promelas*) exposed to various concentrations of effluent. It is well known that Fathead Minnows are particularly sensitive to moderate levels of ammonia. Ammonia interference also caused the artificially high acute toxicities (TUa>1) that were measured prior to 1995, as shown in Table 5.1 of Chapter 5. After 1994, more-accurate measurements of acute toxicity were achieved by reducing ammonia levels in effluent samples prior to testing. In particular, from 1999 through the beginning of 2003, effluent samples were treated with zeolite ion-exchange resin prior to bioassay testing in order to reduce interference from ammonia. However, as described below, this practice was discontinued in April 2003 at the request of the RWQCB staff in their comments on the 2002 Annual Monitoring Report for the MBSCD Wastewater Discharge (RWQCB 2003ab).

The acute bioassay conducted on the April, July, and October effluent grab samples indicated that the projected lethal concentration (LC_{50}), at which 50% of the specimens die after a 96-hour exposure period, exceeded 70.7% (Table 2.7). In terms of toxicity units, this yields a TUa of 1.41, which is only slightly below the permit-specified toxicity limit of TUa =1.5. However, the acute toxicity measured in effluent grab samples collected in 2005 were markedly higher than those measured between 1994 and 2002. This apparent increase in toxicity is not due to any change in the characteristic toxicity of the effluent itself, but is instead an artifact of changes in the protocols used to assess acute toxicity. These changes in protocol were specified by the RWQCB staff and were instituted following the January 2003 bioassay. The new protocols introduced an artificial increase in measured acute toxic concentrations. In contrast, the protocols for chronic bioassays remained the same throughout 2005, as did the measured chronic toxicity.

The recent artificial increase in measured acute toxicity resulted from ammonia interference. During the tests, complex chemical changes within the test solutions caused the effluent ammonia to become highly toxic to the test organisms. Similar chemical changes do not occur in effluent upon discharge, so marine organisms are not exposed to the same levels of toxicity that were measured in the 2005 effluent samples.

At the pH of seawater and wastewater, most ammonia is in the relatively benign, ionic ammonium form. As a result, peak ammonia concentrations (33 mg/L) measured in the MBCSD effluent were over 24-times lower than the instantaneous limit (804 mg/L) imposed for the protection of marine life (see Section 2.2.7 on Page 2-35). However, because the effluent remains static in test chambers during an acute bioassay, complex interactions with the organic constituents in the effluent cause pH to increase. As pH increases, ammonium is converted into a deionized form of ammonia that is highly toxic to the test organisms. In contrast, the pH of marine receiving waters is stabilized by the highly buffered oceanic carbonate system, so discharged effluent never experiences tangible changes in pH (see Chapter 3). Thus, the ammonia toxicity that is present in acute bioassays is not representative of conditions in marine receiving waters.

Various methods have been developed over the years to reduce the well-recognized artifactual toxicity that results from ammonia deionization in wastewater during bioassay tests. Aeration, addition of acid, and pretreatment of the effluent with zeolite are common practices that have been effective at reducing ammonia interference in tests for decades. In the recent past, ammonia interference in the acute bioassays conducted on MBCSD effluent has been successfully eliminated through the use of zeolite pretreatment. Prior to that, when the Kopperdahl (1976) bioassay protocol was required in the previous NPDES discharge permit, CO_2 and O_2 enrichment were found to mitigate the pH drift.

However, after January 2003, the RWQCB (2003ab) specified the use of a new USEPA test protocol (USEPA 2002) that recommends reduction of artifactual ammonia toxicity solely through the use of renewal methods in which the test chambers are periodically refreshed with new test solutions. The RWQCB staff further directed that there should be no other manipulation of the effluent sample prior to testing. However, implementation of daily static renewal of test solutions is significantly more complex and costly than previous bioassay procedures. More importantly, as is clear from the 2005 bioassay results, daily static renewal falls far short of reducing the significant artifactual toxicity that arises from ammonia deionization.

Because static renewal procedures are only marginally effective in accommodating artifactual toxicity resulting from pH-induced ammonia interference, significant effort was expended attempting to reduce impacts from ammonia during the tests. The new bioassay protocols were implemented using daily deliveries of fresh effluent samples from the MBCSD WWTP. Extra care was taken to remove all headspace in sample containers and to maintain low sample temperature during transit to the bioassay laboratory. As recommended in the protocols, sodium thiosulfate was added to some of the effluent samples to neutralize residual chlorine. Although this reduces chlorine toxicity, sodium thiosulfate itself can introduce an artificial increase in measured toxicity. Finally, care was also taken during renewal to avoid physical impacts to the larval fish specimens. Renewal of the test waters can introduce artifactual toxicity from the thermal and physical stress experienced by test organisms during the exchange process.

Despite these efforts, ammonia interference continued to be significant during the acute bioassays conducted in 2005. Its undue influence was evident from the pH and dissolved-oxygen concentrations measured in the test chambers throughout the assays. Ammonia toxicity that was an artifact of the test procedure resulted in the artificially high acute toxicities reported in Table 2.7. The reported results imply that the effluent acute toxicity during the April, July, and October 2005 tests was only 0.09 TUa below the permitted limit, or 94% of the permitted toxicity. In reality, the effluent discharge does not result in pH-induced ammonia toxicity, DO depletion, elevated residual chlorine, sodium thiosulfate, or repeated physical stress on marine organisms upon discharge into receiving waters. Consequently, the reported results are not representative of the toxicity that is experienced by marine organisms exposed to MBCSD effluent.

Moreover, the acute toxicity limitation specified in the current NPDES permit is outdated and unreasonably stringent. The measured acute toxicity concentrations, even when elevated by toxic ammonia interference, would easily meet the limits imposed by the current COP. The most recent version of the COP implicitly acknowledges problems associated with ammonia interference, and the inconsistency in acute toxicity limits that are imposed at the point of discharge rather than at the margin of the initial dilution zone.

In lieu of the much more sensitive chronic bioassay, the COP no longer requires acute bioassays of discharges with dilution ratios less than 1000:1. Because the critical initial dilution of the MBCSD discharge is 133:1, MBCSD would not be required to conduct acute bioassays under the new COP. Even if it were, the revised acute toxicity limits in the current COP would establish a daily maximum acute toxicity concentration of 4.3 TUa for the MBCSD discharge. This is nearly three-times higher than the current limit of 1.5 TUa listed in Table 2.7. Under this revised limit, the highest acute toxicity reported in 2005 (1.41 TUa), despite being artificially inflated due to ammonia interference, would still be well below the new limit. It is incongruous to require new bioassay protocols that were not in place when the NPDES discharge permit was issued to MBCSD without applying improved toxicity limits that are specified in the current COP. In either regard, the actual toxicity of the MBCSD discharge to marine organisms is far lower than is indicated by a comparison of reported and permitted toxicity concentrations in Table 2.7.

2.2.11 Pathogens

In a letter commenting on the 2002 MBCSD Annual Report, the RWQCB staff (2003ab) directed the MBCSD to conduct tests on a set of effluent samples for the presence of *Toxoplasma gondii*. *T. gondii* is a protozoal parasite present in cat feces that may be involved in the increased sea otter (*Enhydra lutris*) mortality found in and around Estero Bay (Miller et al. 2002). However, direct tests for the presence of *T. gondii* oocysts within wastewater, or any other water sample, are not currently available (Conrad 2003). Furthermore, such tests, even if they were feasible, would not provide reliable data for several reasons. First, discharge from the wastewater outfall is an unlikely source of the oocysts. Miller et al. (2002) have clearly demonstrated the overwhelming influence of non-point source contamination and the lack of influence from wastewater discharges; specifically, "*seropositivity to T. gondii was not significantly associated with…proximity to sewage outfalls (P=0.955) but was highly correlated with freshwater flow (P<0.001)*."

Second, even if a test for the presence of *T. gondii* organisms within discrete water samples could be conducted, the incidence of disposal of contaminated cat feces into the collection system would be extremely rare and highly intermittent. *T. gondii* organisms are excreted in cat feces for only two to three weeks, and most healthy exposed cats shed oocysts only during a single brief acute infection stage. Given the infrequency of potential contamination events, it is likely that an excessively large number of discrete samples would have to be collected before measurable oocyst densities could be determined.

Instead, the California Department of Fish and Game (CDFG 2003), in collaboration with the University of California at Davis, has embarked on another approach based on measuring tissue burdens of many types of protozoal and bacteriological pathogens in caged California mussels (*Mytilus californianus*). These mussels have been deployed at selected sites throughout the region. Mussels act as sentinels for pollutants because they bioaccumulate toxins, thereby integrating the affects of highly intermittent contaminant exposure. They also allow detection of very low-level contaminant concentrations that are otherwise imperceptible by even the best tests performed on water samples.

The MBCSD outfall was one of the sites selected for deployment of caged mussels. At various times throughout 2003 and 2004 bags of mussels were deployed on the outfall buoys marking the diffuser structure. The mussels were allowed to filter-feed for a minimum of 30 days after outplanting. Thirty mussels were retrieved from each deployment at the outfall, during the early dry season, late dry season, early wet season, and late wet season. *T. gondii* organisms were not detected in any of the 120 mussels recovered from the outfall buoy (Conrad 2004a). In addition, mussels recovered from the outfall in 2003 were tested for a suite of other bacterial pathogens using enrichment broths and selective media (Conrad 2004b). Mussels collected near the outfall were negative for all bacteria, including: *Campylobacter, Clostridium perfringens, Plesiomonas shigelloides, Salmonella, and Vibrio* spp. (*cholerae, parahaemolyticus*, etc.).

The tests on mussel tissue extend the bacterial tests that are regularly conducted for coliform in effluent (Section 2.2.9) and surfzone samples (Section 3.3.4). They demonstrate the general absence of a wide range of bacterial pathogens, including *T. gondii* organisms, within the MBCSD effluent. The investigators expressed their great appreciation for the MBCSD's financial and logistical support for their research (Conrad 2004a).

2.2.12 Chemical Contaminants

Effluent composite samples were analyzed semiannually in January and July for the presence of 145 chemical contaminants, including trace metals, chlorinated and non-chlorinated phenolic compounds, volatile organic compounds, organochlorine pesticides, PCBs, cyanide, and base-neutral compounds. Chemical analysis of the July sample encompassed an additional 21 compounds, including radionuclides, organophosphorus pesticides, asbestos, and dioxins. Only eleven of the total 167 compounds were present in quantifiable amounts within the effluent samples collected during 2005. The measured concentrations for all eleven compounds were well below the permitted limits. Annual mass emissions of these compounds were also within the goals set forth in the reporting provisions of the discharge permit.

Eleven additional compounds were detected in the July 2005 sample, but at concentrations that were too low to be reliably quantified. Specifically, the measured concentrations of these compounds were above the Method Detection Limit (MDL) but below the Practical Quantification Limit (PQL). These detected but not quantified concentrations were first reported in the analysis of the July 2005 sample, while analysis of prior samples, including the January 2005 sample, only reported quantifiable concentrations above the PQL. Reporting of detected but not quantified concentrations is not required in the current NPDES discharge permit issued to the MBCSD. However, the latest version of the COP (SWRCB 2001) expands the chemistry reporting requirements, and it is likely that subsequent discharge permits issued to the MBCSD will require reporting of detectable concentrations even though they cannot be reliably quantified.

Table C.1 in Appendix C provides a summary of the chemical analyses conducted on effluent samples, along with the corresponding concentration limits specified in the NPDES discharge permit. The laboratory data sheets, pertinent QA/QC data, and chains of custody for the chemical analyses were provided in the semiannual effluent reports (MRS 2005gh). Concentrations for detected compounds are highlighted in bold in the Table. The Table also reports quarterly flow totals, bioassay results, radioactivity measurements, phenolic-compound concentrations determined from grab samples, and ammonia concentrations that were measured on a monthly basis. The compounds with quantifiable concentrations included five trace metals (arsenic, copper, lead, selenium, and zinc), cyanide, a common phthalate compound (bis 2-ethylhexyl phthalate), radionuclides, and dioxin. Two other common chemical constituents, ammonia and chlorine, were also present in effluent samples, as discussed in Sections 2.2.7 and 2.2.8.

Trace Metals

Most metals were not found in effluent samples despite low quantification levels. In Table 2.8, compounds with concentrations too low to be quantified are indicated by the 'less-than' (<) signs preceding the reported PQL. The five metals whose concentrations were quantified in the 2005 effluent samples were found in at least a quarter of the effluent samples collected over the last decade. In all cases, the concentrations measured in 2005 were below the average of concentrations measured in the past. Thus, their presence in the 2005 samples is not indicative of a new contaminant source within the collection system.

Table 2.8 Effluent Trace-Metal Concentrations

	Concentration (μg/L)			Mass Emission (kg/yr)	
Metal	**Limit[a]**	**January**	**July**	**Goal**	**Measured**
Antimony	160800[b]	<100	<100	285	<105
Arsenic	670	**2.8**	<2	17	<4.2
Beryllium	4.42[b]	<10	<10	28	<17.3
Cadmium	130	<10	<10	88	<17.3
Chromium	270	<10	<10	93	<10.8
Copper	140	<10	**15**	690	21.4
Lead	270	**1.6**	**2.1**	465	3.2
Mercury	5	<0.2	<0.2	1.4	<0.35
Nickel	670	<10	<10	142	<13.3
Selenium	2010	**3.0**	<1	65	<3.5
Silver	70	<10	<10	28	<17.3
Thallium	1880[b]	<1	<1	285	<1.0
Zinc	1620	<50	**56**	244	<91.5

[a] 6-month median unless otherwise indicated [b] 30-Day Average

Unlike synthetic organic compounds, trace metals occur naturally within the mineralogy of sediments along the central California coast. Six of the metals listed in Table 2.8 are present in measurable concentrations in seafloor sediments, as described in the benthic chemistry section of this report (Section 4.2). These include four of the metals detected in the effluent (see Table 4.6 in Section 4.2.1). The other metal (selenium) is not tested for in seafloor sediments, although it is also known to be a component of the natural mineral deposits in the region. Thus, the presence of detectable concentrations of all five metals within effluent samples is not unexpected based on inflow of naturally occurring sediments into the collection system. However, copper, lead, and zinc can also enter the collection system through internal corrosion of household plumbing systems.

Selenium has not been detected in effluent samples since July 1998. Anthropogenic sources of selenium are primarily glass manufacturing, fossil fuel combustion (coal burning), and metal refining byproducts. However, the unusual presence of measurable selenium within the MBCSD effluent is more likely the result of the mobilization of naturally occurring selenium in surficial soils following the heavy rainfall event (E2) that occurred immediately prior to the January 2005 effluent sampling. Studies in the San Joaquin Valley watershed have shown that selenium is released from naturally occurring rocks by oxidation during weathering, and that it accumulates in soils as water-soluble salts. In addition, areas where selenium accumulates to elevated levels within soils generally have mean annual precipitation of less than 12 inches. The Estero Bay watershed typically receives less than 12 inches of rainfall annually, but in the three weeks prior to the January 2005 effluent sampling event, over 4.3 inches of precipitation were recorded at the WWTP. It is likely that this intense rainfall mobilized much of the selenium that had accumulated in surficial sediments over the past several years. In any regard, the measured concentration of selenium was 670 times lower that the limit specified in the NPDES discharge permit, and estimated mass emissions for selenium were well below the permitted mass loading goal of 65kg/yr.

Regardless of their source, the quantifiable concentrations of the four remaining metals were also low, and were well below the limitations specified in the NPDES discharge permit. Estimated mass emissions for these trace metals were also well below their respective emission goals. Detectable concentrations of copper have been found in over 85% of the effluent samples collected over the last decade. The presence of detectable concentrations of arsenic, lead, selenium, and zinc in the 2005 samples is somewhat more unusual, with historical occurrence frequencies ranging between 26% and 39%. However, a recent change in the analytical method for some of the metals resulted in lower PQLs for lead and selenium than in most previous analyses. Therefore, detection of these two metals in 2005 may be an artifact of this recent reduction in the detection limit, rather than any increase in concentration or frequency of occurrence. Lead, and selenium may have always been present in effluent samples at concentrations comparable to those reported in 2005, but the analytical method was unable to quantify them until now.

Synthetic Contaminants

Table 2.9 summarizes the remaining pertinent analytical results for tests performed on the two effluent samples collected in 2005. As with the trace metals, the measured concentrations for these constituents were below applicable NPDES discharge limits; in most cases, they were several orders of magnitude lower. Nevertheless, all of these compounds have been occasionally detected in effluent and biosolids samples collected in prior years, and some discussion of potential sources is warranted. Each detected chemical is discussed below.

Bis 2-Ethylhexyl Phthalate (BEHP)

This common phthalate compound has been detected in 75% of the effluent samples collected over the last decade, and was present in both the January and July effluent samples collected during 2005. Phthalate esters, such as BEHP, are components of synthetic dyes, resins, plasticizers, insecticides, and, pharmaceuticals. Nearly 2.7 million metric tons (6 billion pounds) of phthalate esters are produced each year, of which more than half is BEHP. BEHP is a physical plasticizer that is added to plastic resins to soften them, providing increased flexibility. It is not, however, covalently bound to the resin, which allows it to slowly leach out of the plastic and into the environment over time through evaporation or dissolution. Because of their mobility, high vapor pressure, and the massive scale at which they are produced, phthalate

Table 2.9 Quantifiable Concentrations of Other Constituents within Effluent Samples

	Concentration[a] (μg/L)			Mass Emission (kg/yr)	
Constituent	**Limit[b]**	**January**	**July**	**Goal**	**Measured**
Bis 2-ethylhexyl phthalate	470	**6.6**	**10.0**	320	14.2
Cyanide	130	**22**	**21**	57	37.3
Dioxin[e]	0.52	—[d]	**0.159**	1.42×10^{-6}	2.75×10^{-7}
Radioactivity - α (pCi/L)	15.0[g]	—[d]	**0.46±0.89**	—[c]	—
Radioactivity - β (pCi/L)	50.0[g]	—[d]	**11.6±3.01**	—[c]	—

[a] 24-Hour Composite Sample [b] 30-Day Average [c] Goal not specified
[d] Testing not required [e] TCDD equivalents in pg/L

esters in general and BEHP in particular, have become pervasive in the environment. However, although this phthalate compound can negatively affect human health, concentrations would have to be 47-times higher than those measured in July 2005 to be of a human-health concern.

Cyanide

Cyanide was detected in both the January and July effluent samples at concentrations only slightly above its PQL of 20 μg/L. The measured concentrations were almost 6-times lower than the permit limitation and therefore, not of ecological concern. Although cyanide is not normally detected in effluent samples, it is commonly found in low concentrations in biosolids. Although there are many natural sources of cyanide, it is frequently used in pigments and can be introduced into the collection system during photo processing and newspaper printing. Cyanide is also a component in many insecticides. Regardless, at flow rates comparable to 2005, the mass emission of cyanide remained well below the stringent mass-loading goal of 57 kg/yr identified in the NPDES permit.

Dioxin

Dioxin was another ubiquitous chemical detected in the July-2005 effluent samples. Low concentrations of dioxin have been detected in most of the effluent samples tested over the last decade. Trace amounts of dioxin are released to the environment primarily through emissions from the incineration of municipal and chemical wastes, in exhaust from automobiles using leaded gasoline, and from the improper disposal of certain chlorinated chemical wastes. Additionally, dioxins are chemically stable and do not easily break down into other compounds. Therefore, once produced, dioxins persist in the environment and are found almost everywhere. Extremely small concentrations, less than a part-per-trillion, of various dioxin congeners are often found in effluent and biosolids discharged from publicly owned treatment works where they are formed during the chlorination process. The measured dioxin concentration in the July 2005 sample was below the permit concentration limit and the computed mass emission was below the discharge goal.

Radioactivity

A small amount of radionuclides have always been present in effluent samples. Their presence in the 2005 effluent samples is reflected in the alpha (α) and beta (β) radioactivity levels reported in most samples of sediments in Table 2.9. The presence of radionuclides within effluent samples is not unusual given that they occur naturally in the mineralogy of the region. Nevertheless, the levels of gross alpha (α) and beta (β) radioactivity in the July effluent sample were well below drinking water standards. The maximum contaminant levels, pursuant to the California Code of Regulations, Title 22, §64441 and §64443 (Barclay 1997), are 15 pico-Curies per liter (pCi/L) for gross alpha particle activity, including radium-226 but excluding radon and uranium, and 50 pCi/L for gross beta particle activity. Alpha particle activity arises from natural mineral deposits that enter the collection system through erosion. Beta particle activity arises from radioactive decay in both natural and man-made materials.

2.2.13 Mass Loading

The NPDES Monitoring and Reporting Program identifies goals for limiting the mass emission of contaminants from the MBCSD treatment plant (RWQCB-USEPA 1998b). In contrast to the concentration limits specified in the NPDES permit, the mass-emission goals are not part of the waste discharge requirements. Nevertheless, the MBCSD is required to report mass-emission rates for all effluent constituents that have goals listed in the permit. As required, emission rates for 2005, along with the flow rates used in the computations, are provided in Table C.1 of Appendix C.

However, direct comparisons between the goals and reported emissions can be misleading because of how the emission goals were established. As is apparent from the discussion in the previous section, very few chemical compounds have concentrations high enough to be detected in MBCSD effluent. Consequently, most of the mass-emission goals were established using historical PQLs, or, for those chemicals whose PQLs exceeded the permitted limit, the goal was computed using the permit limit. Consequently, most emission goals are reflective of the capabilities of historical chemical analyses, rather than their potential for environmental impact. For most compounds, almost any measurement of a quantifiable concentration would automatically exceed the emission goal, even if the measured concentration remained well below the maximum allowed concentration listed in the Discharge Requirements. Although this has been the case in prior years, no such instance occurred during 2005.

Different computational methods were used to establish the emission goals, depending on whether the constituent had a quantifiable concentration in samples collected between 1994 and 1998. Only six of the 78 compounds listed in the Waste Discharge Requirements had detectable concentrations in effluent samples collected during that period. Their computed mass emissions have the greatest credibility because they were computed from actual measurements, namely, the 99^{th} percentile of their historical concentrations, and a design flow of 2.06 MGD. These compounds are denoted by footnote '13' in the emission-goals listed in the Monitoring and Reporting Program (RWQCB-USEPA 1998b). However, these emission goals are based on past effluent concentrations, rather than on potential impacts to the environment. Such impacts are more-accurately reflected by the much-higher concentrations listed in the NPDES discharge permit. Nevertheless, for these six compounds, a mass loading that exceeds its goal has been deemed '*indicative of statistically significant increases in loading and will trigger the need for antidegradation analysis in the following permit cycle*' (RWQCB-USEPA 1998b). The six compounds are arsenic, copper, zinc, chloroform, halomethanes, and tetrachloroethene. During 2005, none of these compounds exceeded its emission goal (see Table C.1)

The mass-emission goals for the remaining compounds with historical concentrations that were unquantifiable were established using either the maximum allowed concentration listed in the Waste Discharge Requirements (RWQCB-USEPA 1998a) or the historical PQL, whichever was smaller. Because no actual measured concentrations were used to establish the emission goals for these compounds, comparison with actual mass loading in any given year lends no insight into the statistical significance of deviations from the goals, or into the potential for ecological impacts.

Between 1994 and 1998, 15 of these compounds had PQLs that were greater than the maximum allowed concentration listed in the Discharge Requirements. They are denoted with a footnote '*m*' in Table C.1.

For these compounds, the effluent discharge limitation was too low to be measured by standard analytical techniques available at the time the goals were established. As a result, almost any detectable concentration will violate the concentration limits established in the NPDES discharge permit. However, for these compounds, it is inappropriate to compare a mass emission computed from the PQL with a goal computed from the permit limit. Specifically, seven of the 15 compounds had maximum estimated emissions, computed from the PQL, that appear to exceed their goals. They are denoted with footnote '*n*' in Table C.1. However, comparing the projected maximum emission with the goal is misleading because the goals were established using unattainable detection limits. For these seven substances, exceedance of emission goals is indeterminate and their reported maximum emission does not lend insight into treatment-plant performance or the potential for marine impacts. Therefore, as described in the Monitoring and Reporting Program requirements, these seven instances are not subject to antidegradation analyses described above.

2.3 BIOSOLIDS

The monitoring and reporting requirements of the NPDES permit (RWQCB-USEPA 1998b) stipulate that the biosolids be characterized in accordance with 40 CFR 503 (USGPO 1997b). To that end, this section describes the disposition of biosolids generated by the WWTP during 2005. Part of this discussion also addresses the major constituents within the sludge produced by the plant, because they determine the suitability of biosolids for future composting and land application.

Annual biosolids production for 2005 was approximately 225.8 dry metric tons. This annual production was higher than in most previous years because of the preparations for the cleaning and inspection of digester #3. The majority of the 2005 production, 164.2 dry metric tons, was transported to McCarthy Family Farm's San Joaquin Composting facility on December 12th and 13th. An additional 75 dry metric tons were composted at the treatment plant. Of these 75 tons, 61.6 tons were from biosolids produced in 2005, while the remaining 13.9 dry metric tons were from biosolids that remained in storage at the end of 2004.

Solids removed by the primary clarifiers (Figure 2.3 on Page 2-14) are processed as shown in the schematic of Figure 2.16. Sludge is stabilized using two of the three available digesters. It is heated to 36°C and mixed in a primary digester that is coupled in series with a secondary digester. Solids settle in the secondary digester and the supernate is cycled back through the treatment process. The primary digester's capacity is 192,000 gallons, and the secondary digester's capacity is 166,000 gallons, giving a total capacity of 358,000 gallons (1.36 megaL). The volume of sludge pumped to the primary digester from the clarifiers averages 8,000 gallons per day. Average detention times are 44 days. Digester temperature and flow are recorded daily.

Stabilized sludge is drawn from the secondary digester to one of 12 sludge-drying beds, each with a capacity of 5200 ft^2 (483 m^2). Each of these beds has an under-drain and decanting system that re-circulates runoff through the treatment process. Sludge is applied to the beds up to a depth of 12 to 14 inches (33 cm). Drying times typically range from two to four months depending on the weather conditions. Once dried, the biosolids are removed from the beds and stored in a concrete containment area that also drains rainfall runoff through the treatment system. Biosolids are stored in this area until they are removed from the WWTP for composting and eventual use as a soil amendment. Biosolid storage times are generally less than six months.

The biosolids process is highly effective in removing pathogens. This is reflected in the series of pathogen assays conducted on sludge samples that were collected from three different stages of the process in 2002 (Table 2.10). Concentrations of coliform and salmonella were reduced by orders of magnitude in each step of the process, and eventually reached undetectable levels in the composite biosolids sample collected on September 8th from the drying beds. It had no detectable densities of coliform, salmonella, enteric viruses, or viable helminth (parasitic-worm) ova. Thus, the biosolids conformed to the microbial standards necessary to achieve a Class-A status. Class-A biosolids are considered safe for direct contact and unrestricted land application.

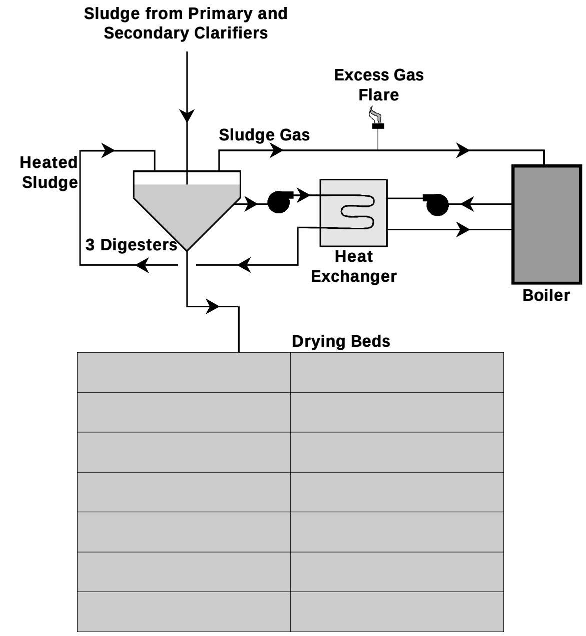

Figure 2.16 Biosolid Process Schematic

Over the past three years, treatment-plant personnel have been developing a beneficial reuse program for biosolids. The program's goal has been to develop and implement a cost effective technique for producing compost that meets the Class-A standards for metals, pathogen reduction, and vector-attraction reduction that are contained in Federal and State regulations. The resulting product meets all the requirements for Exceptional-Quality (EQ) compost that can be beneficially reused in the local community. Ultimately, the EQ compost will be made available to the public for use as a high-quality soil amendment. The composting program utilizes windrow composting similar to that of McCarthy Family Farms, which is described below.

Table 2.10 Pathogen Levels in Different Stages of the Biosolids Process

Stage	Fecal Coliform (MPN/ml)	Salmonella (MPN/4g)
Primary Sludge	160,000	80
Digested Sludge	400	8
Dried Biosolids	<2	<2.2

Biosolids removed from the WWTP in December 2005 were composted at the 162-acre San Joaquin Composting Facility (SJCF) owned by McCarthy Family Farms, Inc. The SJCF is located in Kern County California and operates under Solid Waste Facility Permit No. 15-AA-0287 (Permit Resolution No. 1999-96 adopted by the California Integrated Waste Management Board). The SJCF uses windrow and aerated static pile composting for further reduction of pathogens and limit vector attraction. Windrows are divided into three temperature-recording stations. A windrow passes the high-temperature phase after all three stations maintain a temperature of 55°C for 15 days, and the windrow has been turned at least five times. Six temperature stations are monitored for aerated static pile composting. All six stations must maintain a temperature of 55°C for 3 days before pathogen reduction is considered complete. After maintaining 40°C for 14 days, reduction of vector attraction is also considered complete.

The dried biosolids were sent to the composting facility from the MBCSD treatment plant on December 12th and 13th. Prior to shipping, the MBCSD provided a Class-B-biosolid certification statement based on chemical analyses performed on biosolid samples collected at the treatment plant. Chemical analyses were conducted on a composite sample collected on September 8th from the 182 wet tons of biosolids that were ready for shipment from the treatment plant at the time. Following the requirements of the Monitoring and Reporting Program, the analyses were conducted on a composite of biosolid samples collected from the drying beds. The full laboratory results including chains of custody, instrument calibration reports, and analyses of method blanks and spikes, were reported by MRS (2005b). The results of those chemical analyses are detailed in Table C.2 of Appendix C. They are compared with regulatory limits in Table 2.11.

The tables show that biosolid contaminant concentrations were well below regulatory criteria that would designate them as hazardous, or limit their use for land application or composting. The analyses tested for the presence of 167 potential contaminants, and measured seven other properties and nutrients within the biosolid sample. Three metals (barium, cobalt, and vanadium) that had not been analyzed prior to 2003 were included in the analyte list to provide a complete suite of CAM-17 analytes, even though their analysis is not required under the current NPDES discharge permit. Of the 167 potential contaminants, only 16 chemicals were detected, and 15 of these were naturally occurring trace metals.

All trace-metal concentrations were below Total Threshold Limit Concentrations (TTLC) that would designate the biosolids as hazardous (Table 2.11). Three metals, copper, lead, and selenium had bulk wet-weight concentrations that exceeded ten-times the Soluble Threshold Limit Concentration (STLC). Accordingly, waste extraction tests (WET) were conducted to assess their water solubility. These elutriate tests demonstrated that the metals had low solubility, with dissolved concentrations that were well below STLC levels at which the potential for leaching into groundwater would be of concern. Instead, their low solubility shows that the metals were tightly bound into a mineral matrix, with little bioavailability. The insolubility of trace metals is further confirmed by the low concentrations found in effluent samples (Table 2.8 on Page 2-42). Each of these metals occurs naturally in the mineralogy of ambient sediments in the coast region and, as a result, their presence in biosolids is not unexpected. However, the bulk selenium concentration (31 mg/kg) was somewhat unexpected given historical levels, which typically range between 4 mg/kg and 6 mg/kg. However, as discussed in Section 2.2.12, the higher concentration in 2005 was probably due to the mobilization of naturally occurring selenium in surficial soils after a relatively dry period was followed by a two intense rainfall events (E1 and E2) in early 2005. In any regard, the dry-weight concentrations for all detected metals in the biosolid sample were well less than half of the federally mandated limits, including the monthly limit for materials suitable for agricultural land application (Table 2.11).

Only one synthetic organic contaminant, bis(2-ethylhexyl)phthalate (BEHP), was detected in the biosolids sample. As discussed in Section 2.2.12, BEHP has been consistently detected at low levels in effluent and biosolids samples collected over the past decade. There is no limit on this compound specified in State and Federal regulations governing biosolids. Because of its mobility, high vapor pressure, and the massive scale of production, BEHP has become pervasive in the environment. The other compounds listed in Table 2.11 further characterize the biosolids, as required in the waste discharge requirements. Although testing for asbestos is no longer required as part of the NPDES permit, it was included for completeness. Additionally, Table 2.11 also contains the results of a modified WET test (STLC) for total dissolved solids (TDS) that was conducted in response to a request from the composter.

Table 2.11. Comparison between Measured Biosolid Concentrations and State and Federally Mandated Limits for Hazardous-Waste Designation

		Wet Weight				Dry Weight		
		Measured		Limit		Measured	Limit	
Constituent	Units	Bulk	WET[a]	STLC[b]	TTLC[c]	Bulk	Monthly[d]	Ceiling[e]
Solids	%	90.	—[f]	—	—	—	—	—
Cyanide	ppm	<0.1	—	—	—	<0.11	—	—
Antimony	ppm	<3.	—	15.	500.	<3.3	—	—
Arsenic	ppm	11.	—	5.	500.	12.2	41.	75.
Barium	ppm	570.	—	100.	10,000.	633.	—	—
Beryllium	ppm	<0.2	—	0.75	75.	<0.22	—	—
Boron	ppm	14.	—	—	—	15.6	—	—
Cadmium	ppm	5.5	—	1.	100.	6.1	39.	85.
Chromium	ppm	39.	—	5.	500.	43.3	—	—
Cobalt	ppm	4.5	—	80.	8,000.	5.0	1,500.	4,300.
Copper	ppm	**550.**	10.	25.	2,500.	611.	1,500.	4,300.
Lead	ppm	**56.**	3.6	5.	1,000.	62.	300.	840.
Mercury	ppm	1.3	—	0.2	20.	1.4	17.	57.
Molybdenum	ppm	18.	—	350.	3,500.	20.	—	—
Nickel	ppm	37.	—	20.	2,000.	41.	420.	420.
Selenium	ppm	**31.**	<0.2	1.	100.	34.	100.	100.
Silver	ppm	4.3	—	5.	500.	4.8	—	—
Thallium	ppm	<3.	—	7.	700.	<3.3.	—	—
Vanadium	ppm	21.	—	24.	2,400.	23.	—	—
Zinc	ppm	1,300.	—	250.	5,000.	1,444.	2,800.	7,500.
Asbestos	%	ND	—	—	1.	ND	—	—
Bis(2-ethylhexyl) phthalate	ppm	36.	—	—	—	40.	—	—
Hydrogen-Ion	pH	6.9	—	—	—	—	—	—
Phosphate	ppm	26,000.	—	—	—	29,000.	—	—
Ammonia	ppm	2,000.	—	—	—	2,200.	—	—
Nitrate (NO_3)	ppm	69.	—	—	—	77.	—	—
TDS[g]	ppm	—	3,000.	—	—	—	—	—
TKN	ppm	23,000.	—	—	—	25,600.	—	—
Organic Nitrogen	ppm	21,000.	—	—	—	23,300.	—	—
Oil & Grease	ppm	39,000.	—	—	—	43,300.	—	—

[a] Waste Extraction Tests (WET) measure the soluble leachate or the extractable amount of a substance contained within a bulk sample of biosolids. A WET is indicated if the bulk wet-weight concentration of a contaminant exceeds ten times the STLC.

[b] Soluble Threshold Limit Concentrations (STLC) apply to the measured concentration in the liquid extract from a biosolid sample, as determined by a WET. Biosolids with leachate concentrations exceeding the STLC are classified as hazardous in the State of California, as described in the California Code of Regulations (CCR 2003).

[c] Total Threshold Limit Concentrations (TTLC) apply to the total wet-weight concentration of a contaminant within a bulk biosolid sample consisting of the entire millable solid matrix, rather than just the leachate. Biosolids are designated as hazardous wastes in the State of California if measured bulk concentrations exceed the TTLC, as described in the CCR (2003).

[d] Federally mandated dry-weight limits imposed on biosolids suitable for application on agricultural land apply to monthly average concentrations as defined in Table 3 of the Code of Federal Regulations (USGPO 1997b). [40 CFR §503.13(b)(1)].

[e] Federally mandated dry-weight ceiling concentrations above which biosolids are considered hazardous waste as defined in Table 1 USGPO (1997b).

[f] Measurement not required or limit not specified.

[g] Total dissolved solids determined in a modified STLC extraction

CHAPTER 3

Receiving Water Quality

3.0 RECEIVING WATER QUALITY

This chapter describes environmental conditions within the ocean waters immediately adjacent to the MBCSD wastewater discharge during 2005. The first section presents the details of the monitoring program used to assess compliance with the water-quality objectives of the California Ocean Plan or "*COP*" (SWRCB 1997) as embodied in the NPDES permit applicable to the MBCSD discharge (hereinafter "the NPDES permit," RWQCB–USEPA 1998ab). The second section discusses the disposition of the effluent plume within the receiving waters and how it is influenced by the hydraulic design of the outfall and stratification of receiving waters.

The final section describes the measurements collected during the four 2005 surveys, and compares them with the waste discharge requirements specified in the NPDES permit. The offshore measurements were previously described in quarterly reports (MRS 2005cdef). Quantitative analyses of continuous instrumental measurements and the qualitative visual observations are also described in the last section. All of these observations demonstrate that the diffuser structure was operating as designed, efficiently diluting wastewater within the zone of initial dilution (ZID) that surrounds the outfall. During all four of the 2005 surveys, dilute wastewater was detected within the receiving waters. The slight discharge-related perturbations in the water-property fields indicated that the wastewater was being rapidly mixed upon discharge. Projected contaminant concentrations were far below receiving-water limitations specified in the NPDES permit and the discharge fully complied with the objectives of the COP. The observed perturbations were also small compared to the natural variability in seawater properties that is observed along this portion of the central California Coast. In summary, the highly localized, transient events associated with the effluent discharge were too small in amplitude to be of environmental significance.

3.1 MONITORING PROGRAM

Monitoring of wastewater chemistry (Chapter 2), receiving waters (this chapter), and the biology and physicochemistry of seafloor sediments (Chapter 4) constitute the major components of the MBCSD monitoring program. During 2005, as in the prior 19 years, monitoring of receiving water quality included quarterly offshore surveys conducted near the wastewater discharge within northern Estero Bay, and periodic surfzone coliform sampling along the adjacent shoreline. Measurements collected during the offshore surveys included instrumentally recorded water-property fields, flow velocity determined from drifter trajectories, qualitative observations of general oceanographic and meteorological conditions, and ancillary visual observations of beneficial uses.

3.1.1 Objectives of Sampling

Monitoring of receiving water quality is required under provisions of a 301(h)-modified NPDES permit to determine compliance with the water-quality objectives of the COP (SWRCB 1997) and the Central Coast's Basin Plan (RWQCB 1994). It satisfies the provisions of the Clean Water Act (40 CFR 125.63c) which require a water-quality monitoring program that '… *Provide*[s] *adequate data for evaluating compliance with water quality standards or water quality criteria*….' In addition, water column measurements provide background information on the vertical structure of ambient marine waters near the discharge location. This information aids in estimating minimum initial dilution rates, which are partially determined by the maximum stratification of receiving waters as described in Section 3.2.2 below.

3.1.2 Scope of Monitoring

Many measurements are required to assess whether the water-quality objectives of the COP are being met. These COP objectives are summarized in Table 3.1. They apply to samples collected in an area where initial dilution of the wastewater has been completed.

Initial dilution is the rapid turbulent mixing of wastewater and seawater surrounding the point of discharge. Ambient seawater is forcibly entrained into the discharge plume by two physical processes. Close to the diffuser ports, the momentum of the wastewater jet induces turbulent shear that rapidly entrains near-bottom seawater thereby diluting the discharge. Additional turbulent mixing occurs as the warm buoyant effluent plume rises through the water column. This buoyancy-induced entrainment causes the plume density to approach ambient density (neutral buoyancy) at some point in the water column. Depending on ambient stratification at the time of the discharge, the plume can either reach the surface or become trapped at some intermediate depth level. Thus, initial dilution, where a majority of effluent dilution is achieved, is complete when the diluting wastewater ceases to rise in the water column and first begins to spread horizontally.

To better assess compliance with the COP standards, the receiving-water monitoring program was extensively modified when the current NPDES permit was issued in 1999 (RWQCB–USEPA 1998b). Prior to 1999, the effluent plume was rarely observed in the instrumental measurements and when it was, its lateral extent was largely indeterminate. This was partially due to the pattern of widely spaced stations that coincided with the benthic station pattern. Additionally, collection of discrete water samples for suspended solids, coliform, and oil and grease concentrations was time-consuming and compromised the synoptic nature of the instrumental measurements. Beginning with the 1999 surveys, bottle casts were eliminated, the number of stations was doubled, and the lateral extent of the survey was reduced from 1000 m to 100 m. Rapid instrumental sampling within the tighter sampling pattern resulted in a high-resolution synoptic snapshot of the ocean waters immediately surrounding the outfall. As a result, the disposition of the effluent plume has been accurately delineated in all 28 surveys conducted since the beginning of 1999.

Sampling at the closely spaced stations only became feasible with the increased navigational accuracy of the differential global positioning satellite (DGPS) system. A DGPS system was installed on the survey vessel in 1998 and was used to precisely determine the location of the open section of the diffuser structure during a diver survey (MRS 1998abc).

During the four quarterly surveys of 2005, an automated probe system collected precise *in situ* measurements continuously throughout the water column at each offshore station. Measured seawater parameters consisted of temperature, salinity, dissolved oxygen (DO), acidity/alkalinity (pH), and light transmittance (water clarity). Qualitative visual observations made during the offshore surveys provided information on any potential wastewater contributions to levels of floating particulates, seawater discoloration, odors, algal blooms, and surface water clarity. Wildlife and recreational use, as well as general weather and sea conditions, were also noted.

Table 3.1 California Ocean Plan Water-Quality Standards

A. Bacterial Characteristics

1. Water-Contact Standards

Within a zone bounded by the shoreline and a distance of 1,000 feet from the shoreline or the 30-foot depth contour, whichever is further from the shoreline and in areas outside this zone used for water contact sports, as determined by the Regional Board, but including all kelp beds, the following bacterial objectives shall be maintained throughout the water column:

a. Samples of water from each sampling station shall have a density of total coliform organisms less than 1,000 per 100 ml (10 per ml); provided that not more than 20 percent of the samples at any sampling station, in any 30-day period, may exceed 1,000 per 100 ml (10 ml) and provided further that no single sample when verified by a repeat sample taken within 48 hours shall exceed 10,000 per 100 ml (100 ml).

b. The fecal coliform density based on a minimum of not less than five samples for any 30-day period, shall not exceed a geometric mean of 200 per 100 ml nor shall more than 10 percent of the total samples during any 60-day period exceed 400 per ml.

The "Initial Dilution Zone" of wastewater outfalls shall be excluded from designation as "kelp beds" for purposes of bacterial standards and Regional Boards should recommend extension of such exclusion zone where warranted to the State Board (for consideration under Chapter VI.F.) Adventitious assemblages of kelp plants on waste discharge structures (e.g., outfall pipes and diffusers) do not constitute kelp beds for purposes of bacterial standards.

2. Shellfish Harvesting Standards

At all areas where shellfish may be harvested for human consumption, as determined by the Regional Board, the following bacterial objectives shall be maintained throughout the water column:

The median total coliform density shall not exceed 70 per 100 ml and not more than 10 percent of the samples shall exceed 230 per 100 ml.

B. Bacterial Assessment and Remedial Action Requirements

Describes guidelines for monitoring enterococcus bacteria. (See Plan for full description).

C. Physical Characteristics

1. Floating particulates and grease and oil shall not be visible.
2. The discharge of the waste shall not cause aesthetically undesirable discoloration of the ocean surface.
3. Natural light shall not be significantly reduced at any point outside the initial dilution zone as a result of the discharge of waste.
4. The rate of deposition of inert solids and the characteristics of inert solids in ocean sediments shall not be changed such that benthic communities are degraded.

D. Chemical Characteristics

1. The dissolved oxygen concentration shall not at any time be depressed more than 10 percent from which occurs naturally, as a result of the discharge of oxygen demanding waste materials.
2. The pH shall not be changed at any time more than 0.2 units from that which occurs naturally.
3. The dissolved sulfide concentration of waters in and near sediments shall not be significantly increased above that present under natural conditions.
4. The concentration of substances set forth in Chapter IV, Table B in marine sediments shall not be increased to levels which would degrade indigenous biota.
5. The concentration of organic materials in marine sediments shall not be increased to levels which would degrade marine life.
6. Nutrient materials shall not cause objectionable aquatic growths or degrade indigenous biota.

E. Biological Characteristics

1. Marine communities, including vertebrate, invertebrate and plant species, shall not be degraded.
2. The natural taste, odor and color of fish, shellfish, or other marine resources used for human consumption shall not be altered.
3. The concentration of organic materials in fish, shellfish or other marine resources used for human consumption shall not be bioaccumulated to levels that are harmful to human health.

F. Radioactivity

1. Discharge of radioactive waste shall not degrade marine life.

As in prior years, surfzone monitoring was conducted along Atascadero State Beach to assess aesthetic conditions for the beneficial uses described in Section 2.1.2. Bacteriological conditions along the shoreline are of particular interest for shellfish harvesting and for water-contact recreation, namely swimming and surfing. To assess these conditions, nearshore water samples were collected and analyzed for total and fecal coliform densities. The other beneficial uses, such as fishing and non-contact recreation, such as beachcombing, boating, and picnicking, were addressed by observations of other aesthetic conditions noted during the surfzone surveys. These ancillary observations included weather conditions, ocean currents, tides, and any evidence of water discoloration, floating oil and grease, turbidity, odor, or materials of sewage origin on the beach or in the water. WWTP personnel conducted these surfzone surveys on a weekly basis during the summer and on a monthly basis during the winter.

3.1.3 Sampling Station Design

Water sampling was conducted at 16 offshore stations and eight surfzone stations (Figure 3.1). An additional offshore station (17) was opportunistically sampled during the October survey to characterize the effluent close to a diffuser port shortly after discharge. Surfzone water samples are regularly collected at the eight shoreline stations and are analyzed for fecal and total coliform bacterial densities. These surfzone stations are located at gradient distances upcoast and downcoast relative to Station C, which was positioned at the onshore site closest to the offshore discharge location.

The actual locations of surfzone sampling sites were reevaluated during a high-resolution navigational survey conducted on 6 July 2005. A handheld GPS was used to benchmark the actual locations of the surfzone sites that have been sampled for the past two decades. Historically, the locations of the sampling sites were determined from visual landmarks along the shoreline. Although the locations of the sampling sites have been consistent throughout the monitoring program, their coordinates had never been precisely compared with the target distances specified in the NPDES discharge permit. A comparison of the target and actual along-shore distances is reported in Table 3.2. The detailed shoreline survey, in conjunction with the aforementioned diver survey of the diffuser structure provided a more precise determination of the distance of the discharge from the shoreline. The closest portion of the diffuser structure lies 827 m from the shoreline while the diffuser structure itself extends an additional 52 m toward the northwest along the seafloor. This closest-approach shoreline position was determined at the water's edge when the

Table 3.2 Target and Actual Locations of Surfzone Sampling Stations

Station	Description	Target Along-Shore Distance[1] (m)	Latitude	Longitude	Actual Along-Shore Distance (m)
A1	Upcoast Reference	1700 N	35° 23.966' N	120° 52.113' W	1334 N
A	Upcoast Midfield	610 N	35° 23.744' N	120° 52.060' W	916 N
B	Upcoast Nearfield	305 N	35° 23.520' N	120° 52.002' W	492 N
C	Onshore of Diffuser	0	35° 23.256' N	120° 51.952' W	4 N
D	Downcoast Nearfield	305 S	35° 23.032' N	120° 51.915' W	422 S
E	Downcoast Midfield	610 S	35° 22.762' N	120° 51.895' W	918 S
F	Downcoast Reference	1520 S	35° 22.395' N	120° 51.883' W	1596 S
G	Morro Creek[2]	—[3]	—	—	—

[1] Along-shore distance and direction from Station C as specified in the NPDES permit.
[2] Immediately before flowing to the ocean.
[3] Location varies with the along-shore migration of the mouth of Morro Creek.

Figure 3.1 Locations of Receiving-Water and Surfzone Sampling Stations

tidal level was +2.7 ft, referenced to mean lower low water (MLLW). The beginning of the section of the diffuser structure containing open diffuser ports lies directly offshore surfzone Station C (Figure 3.1).

Seven of the eight surfzone stations were regularly sampled at fixed locations. The Morro Creek Station (G) location, however, varied depending on the along-shore position of the creek outflow. In addition, Station G was only sampled when the creek was flowing to the ocean. Consequently, during 2005, samples were not collected at Station G during portions of the months of September, October, and December when surface outflow from the creek was not observed. Bacterial levels within the creek outflow are monitored because it is a known source of elevated coliform densities that are unrelated to the discharge of wastewater offshore.

Offshore water-quality sampling focused on the water column close to the discharge location (Figure 3.1). Twenty-eight of the 34 available ports discharge effluent along a 42-m section of the diffuser structure. The diffuser ports were hydraulically designed to rapidly dilute effluent within the receiving seawater immediately upon discharge. Most of this turbulent mixing occurs within a zone of initial dilution (ZID), which extends laterally to a distance of approximately 15 m from the center of the diffuser structure.

Beyond the ZID, energetic waves, tides, and coastal currents within Estero Bay further disperse the effluent plume within open-ocean receiving waters. Areas of special concern, such as sanctuaries and estuaries are too distant to be materially affected by wastewater discharge. The southern boundary of the Monterey Bay National Marine Sanctuary is located 38 km to the north, near Cambria Rock. The entrance to Morro Bay lies 2,800 m to the south of the discharge. In 1995, Morro Bay was included in the National Estuary Program. Because of its distance from the discharge, incursion of unmixed wastewater into the Bay is implausible. Furthermore, direct seawater exchange between the discharge point and the Bay is restricted by Morro Rock, which lies between the outfall and the entrance to the Bay. Morro Rock is the largest physiographic feature along the adjacent coastline and extends into Estero Bay approximately 2,000-m south of the point of discharge (Figure 3.1).

During 2005, the discharge's region of influence was evaluated quarterly using water-quality data collected at 16 offshore stations (Table 3.3 and Figure 3.2 on Page 3-7) The stations were situated at three gradient distances from the ZID in order to detect potential discharge-related trends in water properties. The four closest stations, which are on the margin of the ZID, were located 15-m from the closest approach of the diffuser rather than from the center point of the diffuser structure. The remaining, more-distant stations were located relative to the center point of the diffuser structure.

It is important to consider the '*closest-approach*' distance when evaluating impacts at stations close to the wastewater discharge. Although the discharge has historically been considered a '*point source*' for most large-scale modeling purposes, it does not occur at a single location of infinitesimal size. Instead, the discharge is distributed along a 42-m section of the seafloor. The diffuser's finite size affects assessments of wastewater dispersion at sites close to the discharge. Specifically, the amount of wastewater dispersion at a given point in the water column is largely determined by its distance to the closest diffuser port rather than its distance to the center point of the diffuser structure. Near the source, this '*closest-approach*' distance can be much less than the distance to the center of the diffuser structure (Table 3.3, above, and Figure 3.2). It would be misleading to design a nearfield sampling pattern based on center-point distances

Table 3.3 Target Locations of Offshore Receiving-Water Sampling Stations

Station	Description	Latitude	Longitude	Closest Approach Distance[1] (m)	Center Distance[2] (m)
1	Upcoast Midfield	35° 23.253' N	120° 52.504' W	88.4	100
2	Upcoast Nearfield	35° 23.231' N	120° 52.504' W	49.4	60
3	Upcoast ZID	35° 23.210' N	120° 52.504' W	15.0	20
4	Downcoast ZID	35° 23.188' N	120° 52.504' W	15.0	20
5	Downcoast Nearfield	35° 23.167' N	120° 52.504' W	49.4	60
6	Downcoast Midfield	35° 23.145' N	120° 52.504' W	88.4	100
7	Offshore Midfield	35° 23.199' N	120° 52.570' W	85.8	100
8	Offshore Nearfield	35° 23.199' N	120° 52.544' W	46.7	60
9	Offshore ZID	35° 23.199' N	120° 52.519' W	15.0	23
10	Shoreward ZID	35° 23.199' N	120° 52.489' W	15.0	23
11	Shoreward Nearfield	35° 23.199' N	120° 52.464' W	46.7	60
12	Shoreward Midfield	35° 23.199' N	120° 52.438' W	85.8	100
13	Southwest Nearfield	35° 23.176' N	120° 52.532' W	59.8	60
14	Northwest Nearfield	35° 23.222' N	120° 52.532' W	40.2	60
15	Northeast Nearfield	35° 23.222' N	120° 52.476' W	59.8	60
16	Southeast Nearfield	35° 23.176' N	120° 52.476' W	40.2	60

[1] Distance from the closest open diffuser port
[2] Distance to the center of the open diffuser section

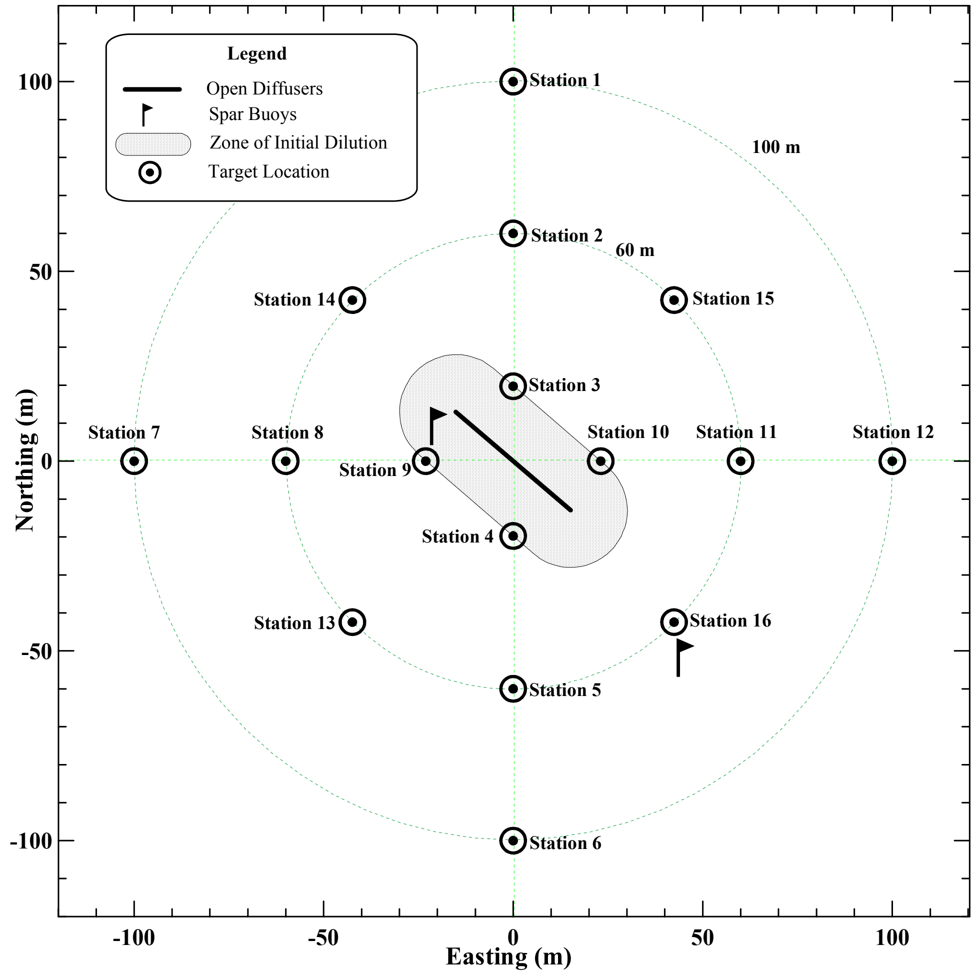

Figure 3.2 Location of Offshore Receiving-Water Stations

rather than closest-approach distances. For example, Station 3 lies on the margin of the ZID, and is potentially influenced by wastewater that is discharged from a diffuser port that lies only 15 m to the southwest. Assuming that mixing occurs throughout a 20-m transit between the center point of the diffuser to Station 3 would overestimate the anticipated amount of wastewater dilution at that site.

With the closest-approach distances in mind, six of the monitoring stations were aligned along a north-south axis at the same isobath (15.2 m) as the diffuser center point. Stations 3 and 4 were positioned at the boundary of the ZID at a distance of 15-m upcoast and downcoast of the closest diffuser ports (Table 3.3). Stations 2 and 5 were located at nearfield distances (60 m) from the diffuser center point. Stations 1 and 6 represent reference stations situated 100 meters upcoast and downcoast of the center point. Depending on the direction of the local oceanic currents at the time of sampling, one or more of these near and midfield stations could conceivably be influenced by wastewater discharge. Under those circumstances, the midfield station on the opposite side of the diffuser acts as a reference station. Comparisons of water properties at these antipodal stations quantify departures from ambient seawater properties so that compliance with the NPDES discharge permit can be evaluated.

Six other stations (7 through 12) were aligned along a cross-shore transect in a pattern matching that of the along-shore transect. The four additional nearfield stations (13 through 16) measure the influence of effluent transported by ocean currents flowing at oblique angles to the bathymetry.

In addition to these 16 regularly sampled stations, a single opportunistic vertical profile was collected within the effluent plume that was visible beneath the sea surface near the diffuser structure during the October survey. The plume was visible because of extraordinarily high clarity of the water column at the time of the survey. The location of this profile, which was directly over the diffuser structure, was designated Station 17. Water-quality sampling at this location is not required as part of the monitoring program but was collected to take advantage of an opportunity to measure the details of the discharge plume while it was in the process of mixing with receiving waters shortly after discharge. None of the measurements collected at opportunistic Station 17 were subject to the receiving-water limitations specified in the NPDES discharge permit.

3.1.4 Navigation

Vessel positioning within the compact offshore sampling pattern only became feasible after the advent of DGPS. The accuracy of traditional navigation systems such as LORAN or standard GPS, is typically ±15 m, a span half of the width of the ZID itself. LORAN beacons are no longer maintained in lieu of the more widely available GPS navigation. Also, prior to 2 May 2000, standard commercial GPS receivers were not allowed to be perfectly accurate by law and a built-in-error system called Selective Availability (SA) was encoded into GPS transmissions. SA could introduce a misreading of up to 100 meters, although it altered most measurements by less than 30 meters. After May 2000, SA was turned off and the accuracy of standard GPS receivers improved substantially, with horizontal position errors that are typically less than 10 m. Nevertheless, extreme atmospheric conditions and physiographic obstructions can also cause signals to bounce around, leading to additional errors in position beyond those that are introduced by SA. These other errors are greatly reduced with the Differential GPS (DGPS) system that was first implemented by the U.S. Coast Guard. DGPS incorporates a second signal from a nearby land-based beacon. Because the beacon is fixed at a known location, the position error in the reading from the GPS satellites can be precisely calculated. This correction is continuously transmitted to the DGPS receiver and results in extremely accurate offshore navigation, typically with position errors of less than 4 m.

At the beginning of 1998, the survey vessel F/V *Bonnie Marietta* was fitted with a Furuno™ GPS 30 and FBX2 differential beacon receiver. This navigational system was used on 29 July 1998 to precisely locate the position of the open section of the diffuser structure (MRS 1998b). Mr. Carson Porter, a diver performing the annual outfall inspection, deployed temporary buoys marking the first and last open diffuser ports. The DGPS locations of these buoys were used to establish the new target locations for the offshore monitoring stations listed in Table 3.3.

Knowledge of the precise location of the actual sampling sites relative to the diffuser is crucial for accurate interpretation of the water property fields. During any given survey, the actual sampling locations do not coincide with the exact target coordinates listed in Table 3.3. Winds, waves, and currents induce offsets of several meters during sampling. Prior to the use of DGPS, these offsets could not be resolved by the available navigation. In contrast, during the year-2005 surveys, the actual sampling locations were precisely determined throughout vertical profiling conducted at each station. This is an important consideration because, in every survey during 2005, station offsets caused some measurements to be collected within the ZID (shaded entries in Table 3.4).

The actual distances between the quarterly station locations and the diffuser structure are compared with the target distances in Table 3.4. Ideally, measurements would be made at or beyond the ZID boundary, namely, at distances of more than 15 m from the diffuser structure. However, because of vessel drift, some samples were unavoidably collected well within the ZID, close to the diffuser structure. In every survey, portions of at least one vertical profile were sampled close (<15 m) to a diffuser port. Because of this, some measurements captured the signature of wastewater while it was continuing to undergo rapid

Table 3.4 Closest Approach Distance between the Diffuser Structure and Actual Sampling Locations during the 2005 Water-Quality Surveys

		Closest-Approach Distance[1] (m)				
Station	**Description**	**Target**	**January**	**April**	**July**	**October**
17	Opportunistic[2]	—	—	—	—	**0.1**[3]
3	Upcoast ZID	15.0	**15.4**	**8.4**	**18.1**	**17.8**
4	Downcoast ZID	15.0	**9.6**	**15.8**	**20.6**	**5.3**
9	Offshore ZID	15.0	**13.9**	22.2	**13.3**	32.4
10	Shoreward ZID	15.0	**10.6**	**13.5**	**16.2**	**15.6**
14	Northwest Nearfield	40.2	38.9	48.8	22.6	29.1
16	Southeast Nearfield	40.2	47.2	46.0	44.4	56.6
8	Offshore Nearfield	46.7	46.3	51.4	53.0	45.4
11	Shoreward Nearfield	46.7	51.6	48.7	42.4	44.3
2	Upcoast Nearfield	49.4	52.4	41.8	52.0	51.3
5	Downcoast Nearfield	49.4	53.7	71.5	47.0	55.5
13	Southwest Nearfield	59.8	65.5	61.6	74.9	59.4
15	Northeast Nearfield	59.8	52.4	58.4	60.4	64.7
7	Offshore Midfield	85.8	95.5	96.9	78.8	81.6
12	Shoreward Midfield	85.8	81.8	77.0	83.3	97.0
1	Upcoast Midfield	88.4	83.7	84.1	90.0	86.0
6	Downcoast Midfield	88.4	91.5	100.5	83.4	106.6

[1] Average Distance to the closest open diffuser port.
[2] Station 17 was a vertical profile collected opportunistically within the effluent plume that was visible beneath the sea surface at the time of the survey.
[3] Portions of the bolded and shaded entries were sampled within the ZID.

initial mixing. Dilution rates were determined from these close-in measurements and compared with critical initial dilution ratios determined by modeling during the design of the outfall. Because tests for compliance with the receiving-water limitations in the NPDES permit apply beyond this dilution zone, the measurements collected within the ZID were not subject to many of the COP standards outlined in Table 3.1. Nevertheless, most of the plume observations collected within the ZID were too small to be considered statistically significant, and were below the limits applicable to observations collected outside the ZID.

Vessel drift can be significant even within the minute it takes to complete the water-property profile at each station. For example, during the October survey, the vessel drifted an average of 21.9 m during the descent of the instrument package through the water column. This magnitude of lateral drift exceeds the dimensions of the 15-m ZID surrounding the diffuser structure and confounds the precise interpretation of compliance at the margin of the ZID. In fact, portions of the vertical profiles collected at Stations 3, 4, 9, and 10 often cross the ZID boundary. Vertical profiles which contain measurements collected within the ZID are indicated in bold in Table 3.4. In some cases, the average positions reported in Table 3.4 are located beyond the ZID even though much of the data was collected within the ZID. To be consistent with historical data where a single location was reported at each station, average positions are based on the surface and bottom position fixes measured at the beginning and end of the vertical profile. However, these average positions can be misleading. Specifically, the average positions reported for Station 3 during the January survey, Station 4 during the April survey, Stations 3, 4, and 10 during the July survey, and Stations 3 and 10 during the October survey, would suggest that the vertical profiles for these stations were collected well outside the 15-m ZID. In reality, large portions of the vertical profiles collected at each of these stations were measured within the ZID, where most of the water-quality limitations do not apply.

3.1.5 Sampling Equipment and Methodology

During 2005, the 38-ft F/V *Bonnie Marietta*, owned and operated by Captain Mark Tognazzini of Morro Bay, provided vessel support for all the offshore surveys. Dr. Douglas Coats of Marine Research Specialists was chief scientist. Ms. Bonnie Luke, also of MRS, was second science officer. Captain Mark Tognazzini supervised all vessel and deck-equipment operations during the water-quality surveys. Standard observations for weather, seas, water clarity/coloration, Secchi depth, and the presence of any odors and floating debris were recorded during the surveys. Wind speeds and air temperatures were measured with a Kestrel® 2000 Thermo-Anemometer. These observations were collected during the rapid water-column profiling that was conducted at each station using a CTD instrument package.

Ancillary Measurements

As vertical CTD profiles were being collected at each station, a Secchi disk was lowered through the water column to determine its depth of disappearance. This provided a visual measure of the turbidity or water clarity within the upper water column. The depth of disappearance is inversely proportional to the average amount of organic and inorganic suspended material along a line of sight immediately below the sea surface. As such, the Secchi depth measures ambient light penetration, which can be limited by increased suspended particulate loads from onshore runoff and wastewater discharge. During most surveys, however, reduced light penetration results from increased turbidity within the surface mixed layer that is caused by upwelling-induced primary productivity within the euphotic zone. The depth of the euphotic zone, where most oceanic photosynthesis occurs, extends to approximately twice the Secchi depth. Because Secchi depths are less precise than measurements collected with electronic sensors, these and the other qualitative observations described above, were ancillary to the rapid water column profiling conducted with an automated instrument package configured with extremely sensitive electronic sensors and probes.

At the beginning of each survey, a drifter was deployed at the center of the open section of the diffuser structure. The drifter was drogued at mid-depth (7 m) using the curtain shade design of Davis et al (1982). In this configuration, the drifter trajectory was largely determined by the oceanic flow field rather than surface winds. Beginning with the October 2004 water-quality survey, satellite navigation was added to the drifter, which allowed continuous recording of the precise time and position of the drifter trajectory rather than just the DGPS location at the points of location and recovery. From these trajectory measurements, the average ambient flow velocity during the survey was determined.

Instrumental Measurements

Vertical water-column profiling was conducted using an electronic instrument package equipped with a number of probes and sensors. A Sea Bird Electronic SBE-19 Seacat CTD[1] instrument package was used to collect profiles of conductivity, salinity, temperature, light transmittance, DO, pH, density, and pressure at each station. A submersible pump on the CTD flushed water through the conductivity cell and oxygen sensor at a constant rate, independent of the CTD's motion through the water column. Prior to the surveys, the transmissivity, DO, and pH sensors were calibrated. The pre-cruise calibration for DO was accomplished by immersing the CTD in an aerated temperature-controlled calibration tank. In addition to this oxygen reading at full saturation, a zero-oxygen calibration point was determined by filling the oxygen-sensor plenum with an 8% solution of sodium sulfite (Na_2SO_3). Oxygen calibration coefficients were determined by regression analysis of sensor membrane current and temperature as recommended by the manufacturer (SBE 1993).

Calibration coefficients for the pH (alkalinity) sensor were determined from a linear regression of output voltage after immersion in three separate buffered solutions of known pH. Buffering solutions with a pH of 4±0.01, 7±0.01, and 10±0.02 were used to bracket the range of *in situ* measurements. The Sea-tech transmissometer was air calibrated by fitting the voltages recorded with and without blocking the light transmission path in air, as recommended by the manufacturer (SBE 1989). The revised calibration coefficients determined prior to each survey were used in the algorithms that convert sensor voltage to engineering units when processing the field data from each individual survey.

The last factory calibration of the entire CTD package was conducted after the October 2001 survey. This and all other factory calibrations confirmed the continued accuracy of the temperature, pressure, and conductivity sensors as well as the operational integrity of the transmissometer, oxygen, and pH probes. The DO and pH sensors were returned to the factory in May 2003 for testing and recalibration. Because of increasing temporal drift associated with the aging DO probe, it was replaced with a new DO probe.

Six seawater properties were used to assess water quality during 2005. They were derived from the continuously recorded output from the electronic probes and sensors on the CTD. Depth limitations on the combination oxygen/pH sensor confined the CTD to depths less than 200 m, which is well beyond the maximum depth of the deepest station (Table 3.5). The precision and accuracy of the various probes, as reported in manufacturer's specifications, are also listed in Table 3.5. Salinity (‰) was calculated from conductivity (Siemens/m) measurements. Density was derived from contemporaneous temperature (°C) and salinity data. It was expressed as 1000 times the specific gravity minus one, which is a unit of sigma-T (σ_t). All three of these physical parameters (salinity, temperature, and density) were used to determine the lateral extent of any discharge-related seawater perturbations. Additionally, they defined the layering (vertical stratification) of the receiving waters, which determines the behavior and dynamics of wastewater as it mixes with ambient seawater within the ZID.

[1] CTD stands for Conductivity-Temperature-Depth

Table 3.5 Instrumental Specifications for CTD Profiler

Component	Depth[1]	Units	Range	Accuracy	Resolution
Housing	600	—	—	—	—
Pump	3400	—	—	—	—
Pressure	680	psia	0 to 1000	± 5.0	± 0.5
Depth	—	meters	0 to 690	± 3.0	± 0.3
Conductivity	600	Siemens/m	0 to 6.5	± 0.001	± 0.0001
Salinity	600	‰	0 to 38	± 0.006	± 0.0006
Temperature	600	º C	–5 to 35	± 0.01	± 0.001
Transmissivity	2000	%	0 to 100	± 0.1	± 0.025
Dissolved Oxygen	200	Mg/L	0 to 21.5	± 0.14	± 0.014
Acidity/Alkalinity	200	pH	0 to 14	± 0.1	± 0.006

[1] Maximum depth limit in meters

Data on the three remaining seawater properties were used to further characterize receiving waters and assess compliance with water-quality criteria. They include light transmittance (water clarity), hydrogen ion concentration (acidity/alkalinity – pH), and dissolved oxygen (DO). Light transmittance was measured as a percentage of the transmitted beam of light detected at the opposite end of a 0.25-m path. Increased transmittance indicates increased water clarity and decreased turbidity.

Before deployment at the initial station, the CTD package was held below the sea surface for a six- to nine-minute equilibration period. Subsequently, the CTD was raised to within 0.3 m of the sea surface and the profiling commenced. The CTD was lowered at a continuous rate to the seafloor. Multiple stations were collected during each deployment by towing the CTD package below the water surface while transiting between adjacent stations. Upon retrieval of the CTD, the profile data were downloaded to a portable computer.

Profile plots and data for each parameter were checked for accuracy and completeness in the field. During the surveys, no obvious irregularities were found that would suggest a malfunctioning sensor. CTD data were deemed satisfactory based on the range-acceptability criteria prescribed for waters of the mainland shelf of Southern California (Table 2 in SCBPPFCT 1995). As described in the next section on data analysis, slow temporal drift in the pH probe was removed prior to processing of data. The locations of the seawater measurements were determined by aligning the time stamps on the internally recording CTD data with the digital navigation log.

3.1.6 Data Analysis

In Section 3.3, the distribution of seawater properties measured during each survey is delineated using vertical sections. These vertical sections help identify the presence of water-property anomalies potentially related to wastewater discharge. They also reveal the inherent spatial variability that results from natural oceanographic processes which helps to place the magnitude of discharge-related anomalies in context. Each cross-shore and along-shore vertical section was derived from a transect of six colinear stations (Figure 3.2). Although the sections are useful for identifying the presence of discharge-related anomalies at most stations, compliance with water-quality standards was quantitatively assessed by directly comparing the standards with statistically significant anomalies identified using measurements at all 16 water-quality stations.

The vertical sections cannot be used to directly determine compliance because they overestimate the true lateral extent of the effluent plume in their depiction of the discharge-related anomalies. Because station spacing is comparable to the width of the ZID itself (Figure 3.2), contours in the vertical sections cannot delineate the sharp lateral discontinuities that define the limits of discharge-related perturbations in seawater properties. High-resolution measurements collected with a horizontally towed CTD have demonstrated that discharge-related anomalies are highly localized around the discharge point within the ZID (MRS 2003a). In contrast, the contouring algorithm used to produce the vertical sections applies a uniform gradient between the localized anomaly and surrounding stations, which often lie well beyond the ZID. In the absence of intervening observations, the artificially expanded horizontal dimension of the contours surrounding the anomaly may appear to extend beyond the ZID. In reality, this is purely an artifact of the contouring algorithm. Because of this, the sections should be used only to determine whether discharge-related anomalies are present at an individual station and not to assess compliance based on interpolated values. Instead, each individual measurement is quantitatively assessed for compliance with the applicable receiving-water limitations using statistical hypothesis tests. The tests evaluate the significance of anomalies, which are computed as the difference between each individual measurement and the mean computed from measurements at the same depth stratum at other stations.

Although all the measured parameters met the field acceptability criteria, the pH sensor exhibited temporal drift during the surveys. This easily identifiable drift has been present in measurements collected in prior years and is due to ongoing sensor-equilibration (MRS 2001, 2002, 2003, 2004). Fortunately, the drift can be accurately removed using regression analysis during post-processing of the data. The maximum required pH adjustment was 0.065 standard units, but most of the vertical profiles did not require an adjustment to pH. Nevertheless, without slight temporal detrending at the remaining stations, minute discharge-related pH anomalies, which are described in Section 3.3.2, would not have been evident.

3.2 DISPERSION OF EFFLUENT WITHIN RECEIVING WATERS

Offshore monitoring is conducted to determine whether the wastewater discharge is causing significant impacts on the receiving waters of Estero Bay beyond the ZID. Section 3.3 makes this determination by comparing the water quality at the boundary of the ZID, and gradient areas beyond the ZID, with background water quality. However, the complex interaction between physical oceanographic processes and ambient seawater properties near the outfall makes detection of the relatively minor discharge-related changes in water quality challenging. The significance of potential water-quality impacts are best assessed through a comparison between the amplitude of discharge-induced anomalies and the inherent variability in ambient seawater properties. Statistically speaking, if the amplitude of the effluent anomaly is small compared to the inherent variability in ambient seawater properties, then the impact is not deemed significant.

The first two subsections of this section describe the character of wastewater dispersion. The first subsection discusses the configuration of the diffuser structure and describes how it determines the dimensions of the zone of initial dilution (ZID). The second subsection presents the dynamics of the rapid mixing that occurs immediately after discharge within the ZID.

The last two subsections document oceanographic processes that affect ambient seawater properties within northern Estero Bay. Regional processes cause major fluctuations in the character of the receiving waters near the diffuser. Seasonal changes in the vertical structure of seawater properties directly affect the magnitude of initial dilution, and the ability to detect the presence of dilute effluent.

3.2.1 Zone of Initial Dilution

Treated wastewater from the MBCSD WWTP is discharged into northern Estero Bay through a 1450-m outfall diffuser system (Table 3.6). The existing outfall was constructed in 1982 with an upgraded 0.686-m diameter steel pipe lined and coated with cement mortar. The outfall extends to a water depth of about 15.2 m (50 ft), approximately 0.5 nautical miles offshore of Atascadero State Beach. At its terminus is a multiport linear diffuser consisting of 34 ports, each 5.08 cm in diameter. The ports are spaced 1.52 m apart on alternating sides of the pipe. The current level of discharge from the WWTP only requires the use of 28 of the available 34 ports.

Table 3.6 Hydraulic Characteristics of the Outfall

Parameter	Measurement
Outfall diameter (m)	0.686
Outfall length (m)	
Land	207
Ocean	1,449
Diffuser diameter (m)	0.686
Diffuser length (m)	51.8
Port orientation (° from horizontal)	0
Port diameter (m)	0.0508
Orifice contraction coefficient	0.89
Center point depth of outfall ports (m MLLW)	15.2
Number of ports open	28[1]
Port spacing[2] (m)	1.52
Design flow rate for each port (m^3/s)	0.01

[1] 28 of 34 ports available.
[2] Ports are located on alternating sides of the diffuser. Distance between ports on the same side of the diffuser is 3.04 m.
[3] Flow rate assuming 34 ports open in a total flow of 0.340 m^3/s (7.76 MGD). The outfall design capacity is 0.358 m^3/s (8.17 MGD).

A Zone of Initial Dilution (ZID) surrounds the diffuser structure and spans the region where the most intense turbulent mixing of effluent is thought to take place shortly after discharge. It is assumed to extend beyond the dimensions of the diffuser structure to a horizontal distance equal to the water depth (USEPA 1994). The water depth at the center point of the diffuser structure is 15.24 m (50 ft) measured relative to the mean lower low water (MLLW) tidal datum. Consequently, the ZID forms a vertical cylinder whose horizontal cross-section is an elongated ellipse as shown in Figure 3.2 on Page 3-7. Based on the diffuser specifications, the dimensions of the ZID are 73.15-m (240-ft) in length, 31.2-m (102.4-ft) in width, and 15.24-m (50-ft) in height. The length is twice the water depth (15.24 m) added to the 42.67-m (140-ft) long diffuser section where ports are open. The width of ZID is twice the water depth added to the inside diameter (0.68 m) of the diffuser pipe. Coatings, which add 0.2 m to the outside diameter of the pipe, and the angled diffuser ports, which extend beyond the edge of the pipe, were neglected for the purposes of determining ZID dimensions. The ZID covers an ocean surface area of approximately 3000 m^2.

In reality, the actual initial dilution zone at any given time departs markedly from the idealized vertical cylinder described above. In many surveys, the prevailing current flow carries the effluent plume beyond the boundaries of the idealized ZID, long before it achieves buoyant equilibrium and before initial mixing is complete. Nevertheless, because wastewater is highly diluted well before initial mixing is fully realized, the water-quality objectives of the COP are consistently met, even within the boundary of the idealized ZID.

3.2.2 Critical Initial Dilution Ratio

Immediately after discharge, wastewater dilution is determined by the physical characteristics of the diffuser system and stratification within the receiving waters. This rapid mixing occurs close to the diffuser structure within the ZID. There, the turbulent processes associated with the high-velocity discharge jet emanating from the diffuser ports, and the buoyant rise of the effluent plume through the water column, result in the rapid dilution of discharged wastewater. Subsequent transport and dispersion by regional oceanographic processes further dissipate the effluent, albeit at a slower rate. The dispersion associated with these oceanographic processes is not considered part of initial dilution.

The limiting concentrations for wastewater constituents specified in the NPDES permit (see Chapter 2) were determined by applying the critical initial dilution ratio for the MBCSD outfall to the open-ocean water-quality objectives specified in the COP for individual contaminants (Table B of SWRCB 1997). The critical initial dilution applicable to the MBCSD outfall was conservatively estimated to be 133:1 (Tetra Tech 1992). The following example shows how this dilution factor was applied to determine the wastewater limits that were established for cadmium in the NPDES permit. The COP's water-quality objective for cadmium in receiving waters has an instantaneous maximum concentration of 0.01 mg/L or less (SWRCB 1997). Based on modeling under highly stratified conditions, there will be at least 133 parts of ambient water mixed with each part of wastewater after completion of initial mixing. From the definition of dilution (Fischer et al. 1979), the concentration of a particular contaminant in wastewater is given by:

$$C_e \equiv C_o + D\left(C_o - C_s\right) \qquad \textbf{Equation 3.1}$$

where: C_e = the concentration of a constituent in the wastewater effluent,
C_o = the concentration of the constituent in the ocean after dilution by D,
D = the dilution ratio of the volume of seawater mixed with wastewater, and
C_s = the background concentration of the constituent in ambient seawater.

The background concentration (C_s) for cadmium in seawater is negligible so the wastewater limit is ($D+1$) times the COP objective (C_o). Thus for cadmium, the instantaneous maximum concentration in wastewater should be no more than 1.34 mg/L, or 134 times the seawater objective of 0.01 mg/L. Thus, under the worst-case scenario, wastewater containing 1.34 mg/L of cadmium will dilute to a cadmium concentration of 0.01 mg/L upon reaching the boundary of the ZID, and the receiving water objective will be met. This is how the maximum wastewater limit on cadmium concentration (1.34 mg/L) was established in the NPDES discharge permit granted to the MBCSD.

The foregoing calculation demonstrates how strongly the assumed initial dilution influences wastewater limitations that are specified in the discharge permit. Because a low value (133:1) was imposed for critical initial dilution, conservative wastewater limitations have been established for the MBCSD discharge. As discussed in Section 3.3, direct observations collected close to the diffuser during the year-2005 surveys indicate that dilutions much higher than 133:1 were achieved well within the ZID and below the assumed plume trapping depth of 6.4 m. The original plume modeling that established the 133:1 critical dilution (Tetra Tech 1992) applied an especially stringent vertical stratification that is not representative of the receiving waters close to the outfall. In addition, the model included 34 open diffuser ports when only 28 ports have been open since the outfall began operations more than a decade ago. Table 3.6 on Page 3-14 lists the other hydraulic properties of the outfall that were used in the initial dilution model.

The density profile used to establish the very conservative MBCSD critical initial dilution was collected at an abandoned monitoring station that was located 1000 m south of the diffuser structure. When the density profile was taken, this location was consistently more stratified than the receiving waters close to the diffuser because the thermal discharge from the Duke Energy (formerly Pacific Gas and Electric PG&E) power facility markedly affected water properties. The energy plant was discharging heated water to the ocean surface near Station F next to Morro Rock (Figure 3.1 on Page 3-5). A decade of monitoring prior to 1999 demonstrated that the thermal outflow from the power plant measurably increased sea-surface temperature at the old monitoring station relative to ambient conditions near the MBCSD outfall. Because this monitoring station consistently exhibited anomalous water properties, it was discarded when the revised receiving-water monitoring program was promulgated in the new permit. Nevertheless, the original plume modeling that established the critical initial dilution ratio used a heavily stratified profile from this station. The model predicted that the receiving waters would trap the effluent plume 6.4 m below the sea surface. More recent dilution modeling demonstrated that under the weaker vertical stratification typical of the water column near the diffuser, a critical initial dilution of approximately 200:1 with a plume trapping depth of 1.4 m is more representative of the outfall performance (Lindstrom 1998).

Even with dilutions as low as 133:1 and highly sensitive instrumentation, the signature of the wastewater discharge would be difficult to detect within receiving waters except very close to the diffuser ports. Accordingly, most wastewater-induced perturbations in water quality are imperceptible beyond the ZID, particularly in the presence of the large, natural oceanic variability that exists within northern Estero Bay. Some of the oceanographic processes that transport contaminants and introduce variability in ambient seawater properties are described in the following subsections.

3.2.3 Regional Oceanographic Processes

Although astronomical tides could occasionally influence the flow near the diffuser structure, a much broader suite of complex oceanographic processes determine the disposition of coastal marine waters within northern Estero Bay. Close to shore, a highly energetic wave field occasionally dominates along this section of the central California coast. This wave energy increases the turbulence within nearshore waters and causes significant sediment re-suspension and sand transport along the outfall corridor. Farther offshore, bands of counter currents move ocean waters in alternating directions along the continental shelf. This large-scale along-shore current flow is interrupted by cross-shore tidal currents and by turbulent jets that occasionally form at major promontories such as Point Piedras Blancas, Point Buchon, and Point Arguello. These narrow energetic jets transport materials far from the coast, and are repeatedly observed extending from upwelling centers located south of major coastal headlands in satellite imagery, such as is shown on the cover of this report. In addition to these processes, wind-driven upwelling alters the vertical distribution of water properties within Estero Bay, and contributes to the remarkably high productivity of the marine waters there.

The open ocean flow field largely controls ambient seawater properties within Estero Bay. Beyond the continental slope (>50 miles), the diffuse southward-flowing California Current represents the eastern limb of the clockwise-flowing gyre that covers much of the North Pacific Ocean. Before turning south to form the California Current, sub arctic water is carried along at high latitudes where it is exposed to atmospheric cooling, nutrient regeneration, and precipitation. As a result, the waters of the California Current are characterized by a seasonably stable low salinity (32 to 34‰), low temperature (13°C to 20°C), and high nutrient concentration. Because of the stabilizing effect of the California Current, central-coast seawater undergoes less seasonal variation than surface waters at similar latitudes along the eastern seaboard.

Immediately shoreward of the California Current, along the continental slope and shelf, is a northward flowing countercurrent that carries water from the Southern California Bight. These southern waters are warmer, more saline and less oxygenated than waters farther offshore. This northward-flowing Davidson countercurrent exhibits strong seasonal variability in intensity. Nevertheless, it maintains a sustained northward flow near Estero Bay, despite reversals observed elsewhere along the California Coast (Coats et al. 1991). The seasonal variability in the Davidson Current coincides with large-scale changes in coastal winds. On average, winds blow from the northwest, parallel to the coast. The Davidson Current is strongest when these northwesterly winds relax between December and February. A rapid spring transition to strong northwesterly winds occurs between March and June when the Davidson Current weakens. These strong northwesterly winds also induce upwelling near Points Estero, Buchon, and Sal (Figure 2.1). During the upwelling season, surface water near the coast is transported offshore and is replaced by deep cool, nutrient-rich seawater.

Significant inter-annual (year-to-year) variations in oceanographic and marine biological properties also occur along the central California coast. They also strongly influence the waters of Estero Bay. These large amplitude variations are associated with the El Niño – Southern Oscillation, which cycles at a period of 3 to 5 years (Graham and White 1988). During El Niño periods, such as in 1982/1983 and in 1997/1998, basin-wide changes in the dynamic balance of wind-driven currents result in modified flow patterns along the coastline of western North and South America (Chelton et al. 1982). Changes in the study region include an anomalous strengthening of Davidson-Current outflow from the Southern California Bight. This increased outflow carries warm, saline sub-tropical waters northward into Estero Bay. It also coincides with increased winter storm activity, reductions in zooplankton biomass, and the introduction of marine organisms typically found far to the south. A relatively strong El Niño condition was present during the latter half of 1997 and the beginning of 1998. It was the cause of the substantial increase in rainfall during the early part of 1998 and heavily impacted central-coast ocean waters.

Superimposed on these large-scale oceanic flows are a variety of transient phenomena including intense eddies, swirls, filaments, meanders, and narrow jets of flow. Some of these cross-shore turbulent features are evident in the satellite image on the cover of this report. They are capable of transporting significant quantities of heat, nutrients, and pollutants to offshore waters (Savoie et al. 1991). Tidal currents provide another mixing mechanism for ocean waters in the study region, although they are not responsible for significant net transport. Estero Bay experiences astronomical tides of diurnal inequality, which means that the two daily sets of tidal extrema are of unequal amplitude. Tidal fluctuations of as much as 2.1 m can combine with storm surges of up to 0.3 m. Typically, however, the range is approximately 1 m between mean high and low water at the outfall site.

At shorter periods, shoaling internal waves and surface gravity waves also mix coastal water properties in both the horizontal and vertical directions. This region of the central coast is relatively unprotected and experiences a comparatively high flux of wave energy, especially in winter when intense storm events are prevalent. Under certain conditions, this wave energy impinges on Atascadero State Beach at oblique angles. This generates substantial alongshore transport within a coastal littoral cell between Point Estero and Morro Rock. The wave-generated littoral cell can extend well offshore into Estero Bay and can encompass the MBCSD discharge location. Four primary meteorological sources generate waves in this region: 1) extratropical winter cyclones in the northern hemisphere, 2) northwesterly winds after the spring transition, 3) tropical disturbances offshore Mexico, and 4) extratropical storm swell generated in the southern hemisphere during summer. The first two are the dominant influences on the wave climate along this section of the central California coast. The last two generate swell from the south, which is diminished near the outfall site by the sheltering effects of Point Buchon (Figure 2.1).

Winter storm waves generated by extratropical winter cyclones are often accompanied by locally severe weather along the south-central coast. These extratropical storms are associated with low-pressure systems that develop along the polar front in the North Pacific Ocean and propagate westward toward the central coast. Thus, major wave events often coincide with an increased discharge from the outfall due to heavy rainfall and increased I&I. These storms occur predominantly in winter (December through March). Less than 5% of all major deepwater wave events (Hs>4 m) occur between the months of May and October (Everts Coastal 1996).

With the exception of major storm events, prevailing northwesterly winds are the predominant mechanism for generating waves in the outfall region. These winds dominate during the spring and summer when a high-pressure system is established over the eastern North Pacific Ocean. The winds are highly coherent over this section of the central coast and consequently, can generate wind waves over a large fetch (Chelton et al. 1987). These locally generated waves tend to be of shorter period and smaller significant wave height than those generated by major winter storms.

3.2.4 Ambient Water Properties

Water properties within Estero Bay are governed by wind stress, wave climatology, runoff from land, and large-scale circulation patterns. The twenty-year record of quarterly water column surveys conducted as part of the MBCSD monitoring program provides the best insight into temporal variation close to the outfall. However, insight into the mechanisms that drive this variability can only be gleaned from a larger-scale study, such as the water-quality survey that was conducted in conjunction with an environmental impact report that examined processes throughout Estero Bay (Morro Group 1999). The vertical sections shown in Figure 3.5 (Page 3-20) and Figure 3.6 (Page 3-22) were collected approximately 7 km south of the MBCSD outfall in April 1999. The figures reveal the large-scale features associated with upwelling processes that are prevalent during the spring, summer, and early fall along the central coast. The upwelling event captured in the satellite image shown on the cover of this report demonstrates that these processes were also present during the October-2005 MBCSD survey.

Upwelling is an important feature of this coastal region and is largely responsible for its productive fishery. Upwelling moves nutrient-rich water to the sea surface, which significantly enhances primary productivity (phytoplanktonic blooms) that is otherwise limited by the lack of nutrients within the euphotic zone. Phytoplankton are the foundation of the marine food web and their increased abundance results in the greater diversity and increased biomass found along this section of the central coast.

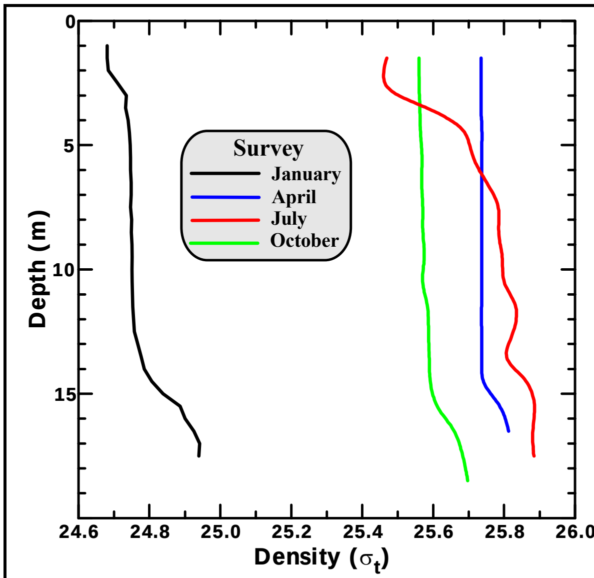

Figure 3.3 Vertical Profiles of Density (σ_t) at Station 7 during the Four Surveys of 2005

Density, Temperature, and Salinity

The density stratification of coastal waters determines the amount of vertical mixing experienced by discharged wastewater as it rises through the water column (Fischer et al. 1979). Highly stratified waters inhibit vertical exchange of nutrients and other water properties, and can reduce dilution of contaminants introduced by seafloor point sources (e.g., ocean outfalls). Vertical stratification varies seasonally as shown in Figure 3.3. During periods when the upper water column is well mixed (weakly stratified), enhanced vertical mixing and increased initial dilution are expected. This normally occurs during the late fall and winter as shown by the black line in Figure 3.3. At that time, convective cooling and mechanical wind stirring drive the main thermocline to great depth (50 m), leaving the nearshore water column with little vertical stratification. However, surveys conducted in April (blue line) and October (green line) 2005 also exhibited only moderate stratification. This was because upwelling also drove the main thermocline to a depth close to the seafloor. As a result, the only perceptible density change was a slight increase below 15 m. In contrast, only the July survey (red line) was conducted during strongly stratified conditions in the upper water column.

Because salinity exerts little influence on density, stratification in marine waters along the central coast is largely determined by temperature structure. Accordingly, Figure 3.4 shows that the vertical temperature structure closely mirrors the density structure shown in Figure 3.3. This was particularly evident during the winter (January) survey, shown in black, when seawater temperatures were highest even though air temperatures are lower in winter. Figure 3.4 shows how seawater temperature near the outfall is controlled by upwelling processes rather than meteorological processes. During strong upwelling, shoreward transport of deep cool seawater resulted in the lower seawater temperatures recorded near the outfall during the April, July, and October surveys (blue, red, and green lines). Reduced sea surface temperatures near the coast during upwelling are also apparent as dark blue regions in the satellite image on the cover of this report.

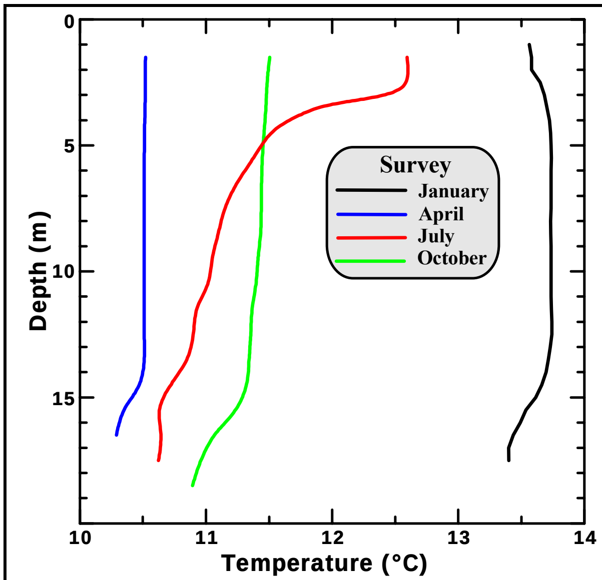

Figure 3.4 Vertical Profiles of Temperature (°C) at Station 7 during the Four Surveys of 2005

Thermal stratification induced by upwelling is also evident in the temperature section of Figure 3.5a in the top frame on Page 3-20. The crowding of isotherms evident between 20 and 60 m resulted from the shoreward transport of cool water near the seafloor. This cool deep water (delineated in blue) moved shoreward, up the sloping bottom to water depths of about 20 m approximately 1000 m from shore. Wave-like oscillations evident in the thermocline arose from internal waves and tides propagating along the interface. These short-period oscillations were captured by the rapid survey, which was completed in approximately 4 hours. Along the transect, the upwelling-induced thermocline also sloped upward toward shore, tracking the movement of deep water.

The shoreward movement of deep cool water during upwelling is also reflected in the salinity and density sections shown in Figure 3.5bc (middle and bottom frames on Page 3-20). The dense near-bottom water mass is more saline, which attests to its origins in the Southern California Bight. The northward flowing Davidson Current brings this saline water into the study region at depth. During upwelling, coastal salinity exhibits a seasonal maximum as a result of shoreward flow at depth.

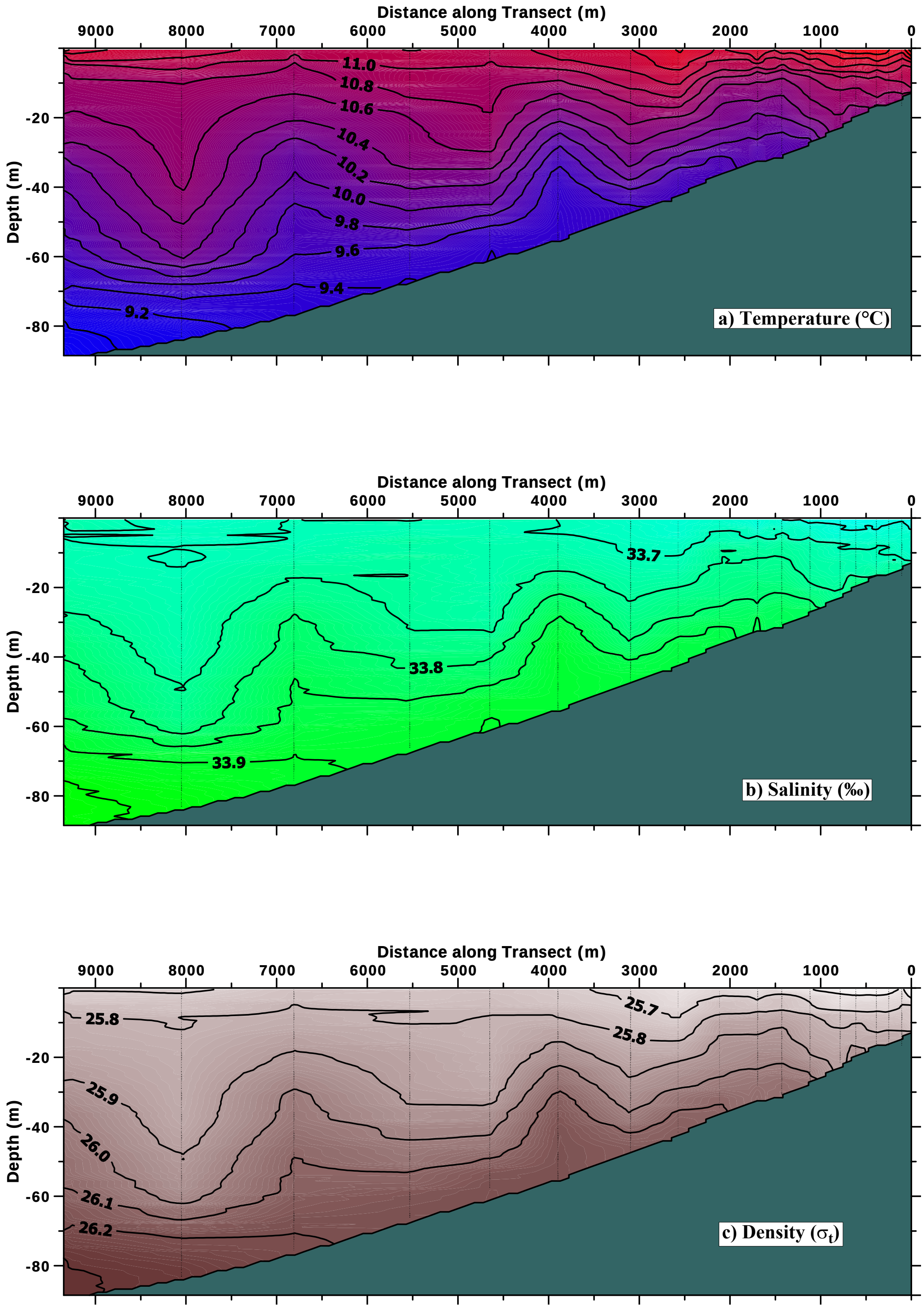

Figure 3.5 Cross-Shore Vertical Sections of a) Temperature, b) Salinity, and c) Density Collected across Estero Bay on 15 April 1999

Dissolved Oxygen

In addition to nutrients, dissolved oxygen (DO) is necessary for a healthy marine ecosystem. Pollutants that are high in organic compounds can locally deplete oxygen levels and can have a deleterious affect on marine organisms. DO depletion arises from the bacterial degradation of oxidizable components in organic wastes. In extreme cases, this additional oxygen demand can reduce DO levels to below those necessary to support biological processes. These anoxic conditions can occur in the water column as well as in seafloor sediments.

In general, however, surface waters are near saturation due to rapid gaseous exchange with the overlying atmosphere. These saturation concentrations are largely determined by sea surface temperature because temperature largely dictates gaseous solubility at the air-sea interface. Below this surface maximum, DO levels steadily decrease with depth due to natural losses from biotic respiration and decomposition, and from the lack of atmospheric exchange except when the water column is well mixed. Ambient DO concentrations within Estero Bay vary seasonally and range from 7 to 9 mg/L near the sea surface. Surface DO levels are typically lowest in the late summer and early fall when sea surface temperatures are highest. This reflects the inverse relationship between oxygen solubility and temperature. Under the stratified conditions during upwelling in spring and summer, DO levels decrease rapidly with depth and decline to 5 mg/L in as little as 50 m (Figure 3.6a, top frame on Page 3-22). These low oxygen concentrations are a consequence of the shoreward movement of deep oxygen-poor water. These deep waters have not been in recent contact with the atmosphere, and ongoing respiration and decomposition have resulted in undersaturated oxygen levels along with enhanced nutrient levels.

pH

The hydrogen ion content (pH) of marine waters in the study region is similar to seawater in other oceans of the world. It is slightly alkaline with a pH ranging between 7 and 8. The lack of strong geographic variation in pH is a consequence of the well-buffered nature of carbonate system in seawater. The highest pH levels occur in the study region during spring upwelling when increased photosynthesis consumes carbon dioxide (CO_2) and produces oxygen near the sea surface. As the ratio of respiration to photosynthesis increases with depth, there is an increase in CO_2 and a decline in pH (more acidic).

Although the range is small, this vertical decrease in pH is evident in Figure 3.6b (middle frame on Page 3-22). Seawater alkalinity can also be temporarily affected by discharge of waste into the ocean if the pH of the wastewater is substantially different from that of the receiving waters. However, because of the well-buffered carbonate system, it tends to have only a localized effect in open-ocean waters.

Turbidity

Turbidity decreases the clarity of seawater and can limit the penetration of ambient light in the upper reaches of the water column. Turbidity is largely determined by the concentration of suspended particulate matter. Turbidity dictates the depth of the euphotic zone. Within the euphotic zone, ambient light intensity exceeds roughly 1% of surface illumination, which is the minimum necessary for phytoplankton growth. Turbidity is increased in coastal waters by phytoplankton blooms, storm runoff, sediment resuspension, and discharge of wastewater from municipal outfalls. Along the central California coast, substantial sediment input from rivers and runoff occurs in the form of large isolated pulses rather than a steady discharge of material. Rare intense storm events occasionally punctuate the prevailing semi-arid climate and result in large runoff with profound but transient increases in coastal turbidity.

The MBCSD monitoring program measures turbidity around the outfall in two ways:

- Continuous *in situ* measurements of transmissivity, which is the relative amount of light transmitted along a fixed path; and
- Secchi-disk depth, which measures ambient light penetration near the sea surface.

Secchi depths range between 1 m and 10 m within a few miles of the coast but extend to 30 m (100 ft) within the California Current farther offshore. Near the outfall, Secchi depths typically range between 2 and 7 m. During the October 2005 survey, Secchi depths up to 10 m were recorded. The euphotic zone extends to approximately twice the Secchi depth. The greatest coastal water clarity in the upper water column and the deepest Secchi depths often occur during the late summer and fall when runoff and upwelling are minimal. During this time, the euphotic zone can extend throughout the entire water column near the coast.

Coastal turbidity is also heavily influenced by wave-induced sediment re-suspension near the seafloor. Immediately outside the surf zone, transmissivity decreases near the seafloor during periods of high wave activity. This phenomenon is often observed at the outfall during the winter. When it coincides with upwelling, turbidity is also high at the surface from the presence of biogenic particulates. As a result, a mid-depth maximum in transmissivity (water clarity) is evident in vertical sections (Figure 3.6c, bottom frame on Page 3-22).

3.3 EFFECTS OF DISCHARGE

This section describes discharge-related trends in seawater properties observed during the four quarterly surveys of 2005. In particular, it evaluates compliance of the year-2005 wastewater discharge with the NPDES permit limitations on water quality. For the most part, the water-quality limitations were based on criteria in the COP (Table 3.1), the Central-Coast Basin Plan, and other state and federal policies that were designed to protect marine life and beneficial uses of ocean waters. These receiving water standards are intended to evaluate whether discharges could cause significant impacts on water quality beyond the zone of initial dilution (ZID). Consequently, a part of this evaluation examines spatial trends that are related to outfall proximity and whose amplitude significantly exceeds the inherent background variability arising from ambient oceanographic processes.

Spatial variability in offshore water quality is determined from the three-dimensional distribution of instrumental measurements of seawater temperature, salinity, density, transmissivity, DO, and pH that were collected on a quarterly basis. For timely reporting of water-quality status, a receiving-water report was prepared after each quarterly survey (MRS 2005cdef). Those reports included comprehensive listings of raw data that are not provided in this annual report. They also included detailed discussions of vessel offset and perturbations observed in the instrumental measurements.

In contrast, the summary presented in this report consolidates the detailed results from each individual survey into an annual context where the influence of large seasonal variability in receiving-water properties becomes apparent. Table 3.7 provides a synopsis of conditions observed during the four offshore surveys. It guides the discussion of the seawater parameters presented in the subsections below. As described above, the ambient receiving-water conditions are largely dictated by the strength of upwelling processes that prevail at the time of the survey. The relative strength of upwelling can be gauged from indices determined from wind measurements collected along the central coast (Figure 3.7).

Table 3.7 Oceanographic Conditions during the Four Quarterly Receiving-Water Surveys

Conditions	January (winter)	April (spring)	July (summer)	October (fall)
Tide	Flood	Slack	Flood	Slack
Wave Height	1/WNW	4/W	2/W	3/NW
Flow Direction	WSW then NE	Southwest	Northeast	Southeast
Stratification	Moderate	Moderate	Strong	Moderate
Water Clarity	Moderate	Low	Low	High
Upwelling	Weak	Moderate	Moderate	Strong
Plume Visible	No	No	No	Yes
Plume Signature	Yes	Yes	Yes	Yes

As shown in the Figure, all of the water-quality surveys were conducted on days when there was some degree of upwelling. The solid line shows the daily upwelling index where positive values reflect the strength of southeastward-directed winds near Estero Bay. These winds drive offshore surface flow, thereby inducing upwelling near the coast. A comparison of the daily index (solid line) and 30-day running average (dashed blue line) reveals substantial daily variation in the strength of the along-shore winds. On average, the strongest upwelling occurred in the latter-half of April 2005. However, only a moderate level of upwelling prevailed during the April water-quality survey both because the survey was conducted in early April, and because on the day of the survey, upwelling winds had begun to relax.

Conversely, the strong upwelling that prevailed during the October survey was due to unusually strong northwesterly winds[2] that prevailed all along the central coast in the early part of the month. The resulting upwelling extended along the entire coast, as is apparent in the satellite image shown on the cover of this

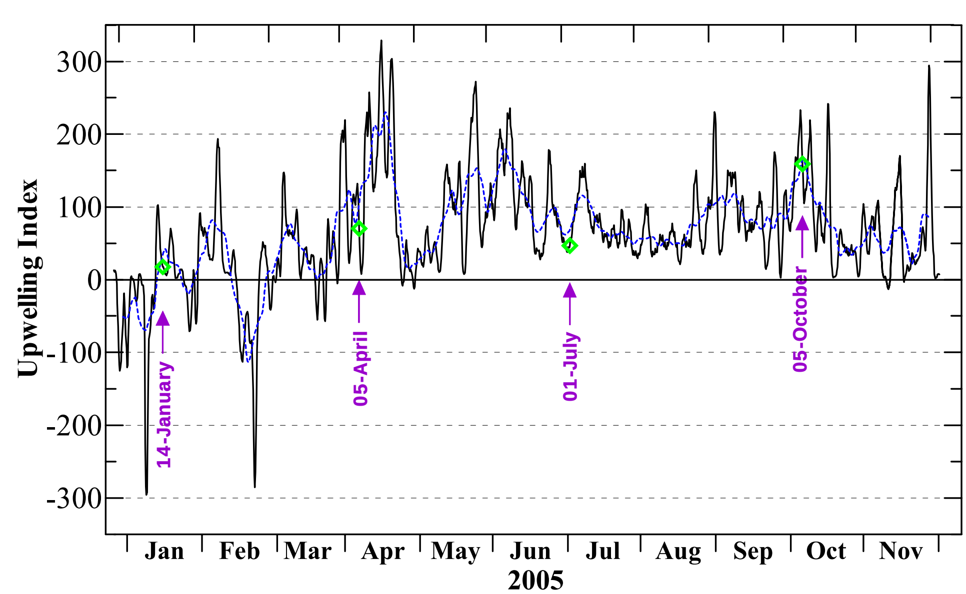

Figure 3.7 Daily and Monthly Average Upwelling Index ($m^3/s/100m$ of coastline)

[2] "Northwesterly" winds refer to winds coming from out of the northwest.

report. The cover image was recorded the day before the survey when skies were clear enough for sea-surface temperatures to be measured by infrared sensors on one of NOAA's polar orbiting satellites. The presence of much cooler seawater near the coast is delineated in the image by blue (13°C) and purple (12°C) shading. At major promontories, this upwelled water extends well offshore. Within Estero Bay, cold sea-surface temperatures near or below 12°C, were consistent with the near-surface temperatures measured by the CTD during the survey, as shown by the green line in Figure 3.4 on Page 3-19. The satellite image shows that because of coastal upwelling, seawater temperatures near the outfall were more than 4°C cooler than ambient temperatures farther offshore.

Although the plume signature was detected by instrumental observations in all four surveys, it was observed visually only during the October survey. Despite visual and instrumental manifestation of the effluent within receiving waters, neither odors nor debris of sewage origin were observed at the sea surface. In fact, all of the instrumental data indicated that water properties beyond the ZID were typical of natural background conditions in coastal waters. As described in the following subsections, only slight salinity anomalies and a possible seafloor turbidity increase were observed beyond the ZID. These anomalies indicated the presence of exceedingly dilute wastewater that was continuing to undergo a minor amount of initial mixing. In contrast, wastewater undergoing intense mixing and rapid dilution was observed very close to diffuser ports. Both of these sets of observations attest to the continued effective and efficient operation of the outfall and diffuser structure. Qualitative observations of aesthetic characteristics and beneficial uses also documented the lack of impacts from wastewater discharge. These aesthetic observations demonstrated compliance with Section C (Physical Characteristics) of the COP (Table 3.1). The few incidents of high coliform densities in the surf zone were correlated with non-point source (onshore) run-off and not with the discharge of wastewater. All other receiving water parameters were also in compliance with permit criteria.

3.3.1 Observations of Beneficial Use

Observations of beneficial use demonstrated that the coastal waters in the outfall vicinity and along adjacent shoreline continued to be utilized by humans and wildlife during the year-2005 water-quality surveys. There was no evidence indicating that the presence of the discharge enhanced or detracted from the beneficial uses that were identified for northern Estero Bay in the Basin Plan (see Section 2.1.2). Specifically, fish, marine mammals, and seabirds were observed utilizing the area within and around the ZID area as marine habitat (*viz.*, the *MAR* beneficial use identified for the waters of Estero Bay in the Central-Coast Basin Plan outlined in Section 2.1.2). The presence of surfers, beachcombers, and commercial and recreational fishing boats attested to the use of the waters for water recreation (*REC-1* and *REC-2*), transportation (*NAV*), and fishing (*COMM*).

A variety of marine organisms were observed during the 2005 water-quality surveys. Occasionally, juvenile sea lions (*Zalophus californianus*) made their presence known during the surveys. Pieces of detached floating kelp (*Nereocystis luetkeana* and *Macrocystis* spp.) were also commonly observed during the offshore surveys. Seabirds, including Brandt's Cormorants (*Phalacrocorax penicillatus*), Surf Scoters (*Melanitta perspicillata*), and California brown pelicans (*Pelecanus occidentalis californicus*), were observed transiting through, and foraging within the survey area. Surf Scoters are common along Pacific Coast in late winter and usually stay some distance from shore, feeding on shellfish. The males are distinguished by a bright red-orange pattern on their bill.

California brown pelicans (*P. o. californicus*), a federally and state listed endangered species, were also often observed in flight near the survey area. They feed by plunge diving from heights of up to 15 to 20 m above the ocean surface, and primarily consume small schooling fish (e.g., anchovies) (USFWS 1982). Pelicans return to specific roosts each day and do not normally remain at sea overnight. These roosts are usually in regions of high oceanic productivity and isolated from predation pressure and human disturbances. Flocks of pelicans thriving near the outfall are noteworthy because pelicans are particularly sensitive to chemical pollutants absorbed from the fish they eat (McConnaughey and McConnaughey 1985). Serious declines in the southern California population prior to 1980 were linked to bioaccumulation of chlorinated hydrocarbon pesticides (DDT, DDE, dieldrin, and endrin) in the pelican's food chain (USMMS 1996). None of these compounds were detected in wastewater discharged by the MBCSD during 2005 (see Section 2.2.12). Several decades ago, at other sites along the California Coast, bioaccumulation of these pesticides resulted in eggshell thinning and poor reproductive success for the brown pelican (Schreiber and Risebrough 1972). Food scarcity, principally anchovies, also contributed to the species' decline in the 1960's (Keith et al. 1971).

In July 2005, southern sea otters (*Enhydra lutris nereis*) were also observed during transit to the survey area from Morro Bay, and one was observed transiting the area 20 m north of the diffuser structure. Their presence attests to area's designation as a habitat that supports '*…animal species that are established under state or federal law as rare threatened, or endangered*' (*RARE*, see Section 2.1.2). Sea otters are a federally and state-protected species. Since the arrival of a raft of 30 to 40 sea otters just north of Port San Luis was observed in 1974 (Gotshall et al. 1984), they have become increasingly prevalent in the vicinity of Estero Bay (Gotshall et al. 1986). Sea otters feed almost entirely on macroinvertebrates (Ebert 1968, Estes et al. 1981, Riedman and Estes 1990) and the southward expansion of the sea otter is firmly correlated, chronologically and geographically, with the demise of the abalone fishery in the region (Morro Group, Inc. 1999). Based on the spring 2005 census, there are currently about 2,735 southern sea otters in the waters off California extending from Half Moon Bay south to Santa Barbara. The majority of male otters make regular long-distance movements throughout this range. The three-year running average of the otter population is currently around 2,700 individuals, which is an increase of about 8% over the previous three-year average. The three-year average population must exceed 3,090 for three consecutive years for the population to be delisted.

Marine-habitat (*MAR*) and ocean-fishing (*COMM*) beneficial uses were also in evidence during the 2005 surveys. For example, sport fishing vessels were observed transiting the survey area and non-contact water recreational activities, designated as *REC-2* in the Basin Plan, were consistently observed along Atascadero Beach during the regularly conducted surf zone surveys. Individuals were documented sunbathing, hiking, jogging, surf fishing, beachcombing, and walking their dogs on the beach. Dog observations were largely restricted to Stations E, G, and F because the northern section of Atascadero Beach was closed to dogs beginning in 2002. Beach camping is permitted at the Atascadero State Beach RV Park located near surf zone Station A shown in Figure 3.1. Water-contact recreation, designated *REC-1* in the Basin Plan, also occurs frequently along the beach adjacent to the outfall. This includes activities such as surfing, swimming, body surfing, kayaking, and kite and wind surfing.

3.3.2 Instrumental Observations

The signature of the effluent plume was detected in all four surveys conducted during 2005. Table 3.8 and Table 3.9 on Page 3-27 summarize the nine identifiable perturbations in seawater properties that were associated with wastewater discharge.

Table 3.8 Perturbations in Dynamical Seawater Properties Associated with Effluent Discharge

Pertur-bation	Survey	Station	Dilution	Anomaly[1] Salinity (‰)	Anomaly[1] Temperature (°C)	Anomaly[1] Density (σ_t)	Depth (m)	Height of Rise (mab)[2]	Diffuser Dis-tance (m)[3]
P1	January	3	≥807[4]	–0.041	–0.20	+0.063	11.0	5.0	14.1
P2	April	4	≥762	–0.044	–0.13	–0.015	3.5	13.5	15.5
P3	April	9	≥413	**–0.081**[5]	–0.10	–0.049	1.5	16.0	20.1
P4	April	13	≥465	**–0.072**	–0.06	–0.046	2.5	15.0	60.5
P5	July	4	≥784	–0.043	–0.16	–0.038	10.5	6.5	21.0
P6	July	9	≥636	–0.053	–0.09	–0.026	8.0	9.0	13.4
P7	October	4	≥429	**–0.078**	**–0.48**	—[6]	8.5	9.0	2.5
P8	October	17[7]	≥346	**–0.097**	–0.37	–0.047	4.0	13.5	0.1
P9	October	17	≥100	**–0.331**	—	**–0.274**	15.5	2.0	2.9

[1] Amplitude of a water-property measurement above the average value computed from measurements at the same depth level at all other stations. For a given perturbation, the largest anomaly is listed.
[2] Height-of-rise in mab (meters above the bottom) to the largest anomaly within the water column, *i.e.* vertical distance measured from the seafloor rather than the sea surface.
[3] Horizontal distance from the largest salinity anomaly to the closest portion of the diffuser structure. Compare this distance with the 15.2-m wide ZID for assessing compliance.
[4] Based on the observed salinity anomaly and Equation 3.2.
[5] Anomalies highlighted by shading and bolding were found to be statistically significant at the 95% confidence level.
[6] No discharge-related anomaly was evident in this water property.
[7] Station 17 was an opportunistic vertical profile collected directly over the diffuser structure and within the effluent plume visible beneath the sea surface.

Table 3.9 Perturbations in Water-Quality Parameters Associated with Effluent Discharge

Pertur-bation	Survey	Station	Dilution	Anomaly Trans-missivity (%)	Anomaly DO (mg/L)	Anomaly pH	Depth (m)	Diffuser Dis-tance (m)
P1	January	3	≥807	**+8.4**	–0.41	**–0.037**	1.0–12.5	14.1
P2	April	4	≥762	+5.0	–0.66	–0.019	1.0–8.0	15.5
P3	April	9	≥413	+2.1	–0.60	**–0.031**	1.0–13.0	20.1
P4	April	13	≥465	+2.2	–0.41	–0.017	1.0–5.0	60.5
P5	July	4	≥784	**–5.3**	–0.39	—	6.5–16.5	21.0
P6	July	9	≥636	—	–0.34	—	7.0–8.5	13.4
P7	October	4	≥429	—	**–1.04**	**–0.048**	1.0–14.5	2.5
P8	October	17	≥346	—	**–0.94**	**–0.045**	4.0–10.0	0.1
P9	October	17	≥100	**–3.7**	—	—	14.0–16.0	2.9

Five of these plume observations (Perturbations P1, P6, P7, P8, and P9) were located within the ZID boundary, where most receiving-water limitations do not apply. All of the perturbations were associated with anomalies in the seawater properties that determine the dynamics of the flow field, namely salinity, temperature, and density (Table 3.8). All of the perturbations were associated with anomalies in the salinity field (Figure 3.8), as well as in one or more of the properties related to water quality, namely, transmissivity, DO, and pH (Table 3.9). All but two of these nine water-quality perturbations were generated by the upward displacement of deep ambient seawater that had been entrained by the rising effluent plume. The only exceptions were the seafloor transmissivity anomalies associated with Perturbations P5 and P9. The decreased transmissivity measured close to a diffuser port at Station 17 during the October survey was caused by the presence of trace amounts of wastewater particulates that were in the process of dispersing within the ZID shortly after discharge.

Two distinct discharge-related processes create anomalies in seawater properties. Both of these physical processes can arise from the seafloor discharge of buoyant effluent.

- Effluent-induced anomalies caused by the presence of trace amounts of wastewater constituents that have yet to completely mix with receiving waters
- Entrainment-generated anomalies caused by the upward displacement of deep ambient seawater that has been entrained by the rising effluent plume

As is discussed below, the character of the seawater anomalies differentiates these two types of discharge-related processes. Often, both processes act together to form a series of vertical anomalies wherein slugs of partially mixed ambient bottom water are carried upward by the buoyant effluent plume. This creates tightly packed vertical gradients in seawater properties that are indicative of ongoing turbulent mixing.

Most of the time, anomalies in seawater properties other than salinity, are generated by the entrainment of ambient bottom water. This is evident from the similarity in the properties associated with the anomalies, and the properties in ambient seawater at depth. The vertical connection between entrainment-generated anomalies and deep ambient seawater is particularly apparent in Perturbation P1, which is shown in the vertical section of Figure 3.10 on Page 3-37. In contrast, wastewater-induced anomalies tend to be spatially isolated from the properties of ambient seawater near the seafloor. This isolation is apparent with most of the salinity anomalies shown in Figure 3.8. Similarly, the highly localized transmissivity anomalies shown in Figure 3.9 on Page 3-35 were recorded close to a discharge jet. They clearly indicate the increased presence of wastewater particulates that have yet to dissipate to concentrations below the detection limits of the highly sensitive CTD instrumentation.

Anomalies Unrelated to Discharge

A number of statistically significant anomalies identified in the 2005 water-quality database were clearly unrelated to the discharge. There were 12,336 water-quality observations reported at standard 0.5-m intervals in 2005. Each observation was evaluated for its departure from mean conditions. For each observation within a given survey, its departure or anomaly was computed by subtracting the observed value from the average of the 15 other measurements collected at the same depth level at other stations. Measurements collected within 10 m of the sea surface were compared with the mean of the measurements collected at the same distance below the ocean surface at the 15 other stations. Deeper measurements were compared with the mean of measurements collected at the same height above the sloping seafloor. This is because deeper water properties tend to parallel the sloping seafloor rather than the horizontal sea surface.

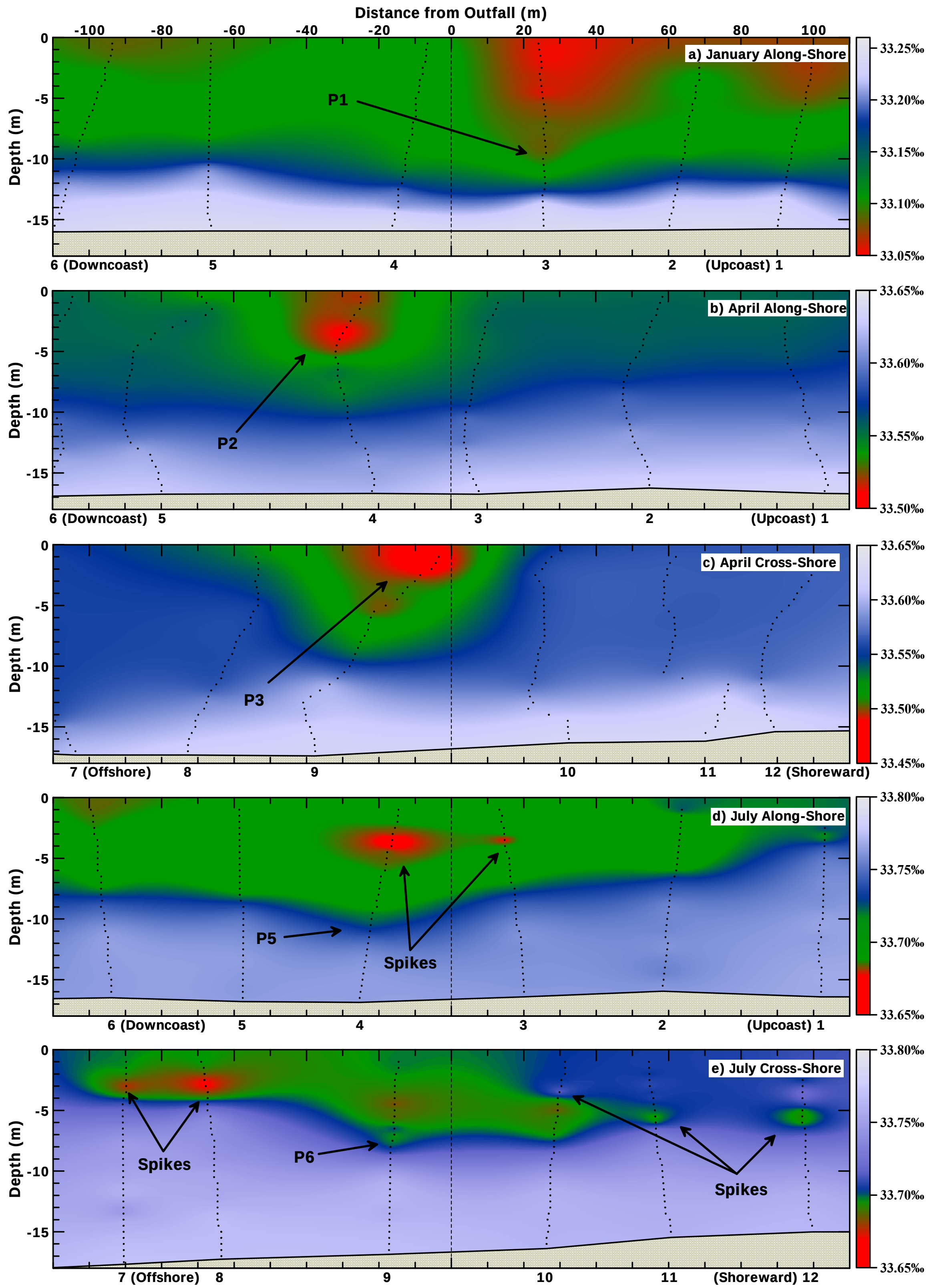

Figure 3.8 Discharge-Related Salinity Perturbations

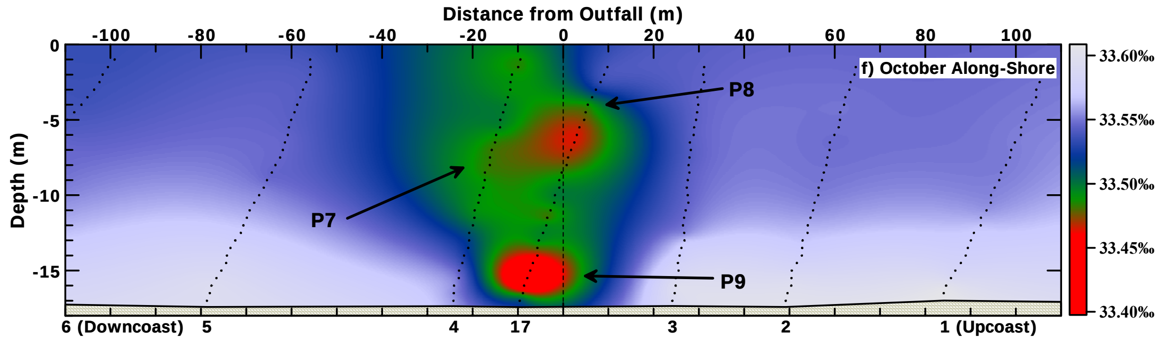

Figure 3.8 Discharge-Related Salinity Perturbations (continued)

The statistical significance of these departures from ambient conditions was evaluated using an analysis of variance that compares the amplitude of a single anomaly with the inherent natural variability in observations (see Page 228 in Sokal and Rohlf 1997). Natural variability was estimated from the standard deviation of all measurements for a given seawater parameter (e.g., salinity). The COP (SWRCB 1997) defines a '*significant*' difference '*as a statistically significant difference in the means of two distributions of sampling results at the 95 percent confidence level*.' Consequently, statistically significant anomalies were identified as those that departed from mean conditions by more than the 95-percent confidence interval determined from the standard deviation and number of observations used to compute the average.

Although 15 independent hypothesis tests were performed at each depth level, no Bonferroni adjustment to the error rate was included; so the tests are conservative. As a result, the actual confidence level for the overall null hypothesis test for differences was higher, approximately 99.7%, rather than the 95% level that applies to a single test. Also, the standard deviation that was applied in the tests was determined from the entire data set collected during a given survey. Thus, it reflects the full range in the variability in ambient properties, including vertical variations. As a result, an anomaly of a given amplitude is less likely to be found statistically significant if it was measured at a time when the water column was highly stratified, than when the water column was relatively unstratified.

Although about one-third of the discharge-related anomalies were statistically significant (bolded and shaded in Table 3.8 and Table 3.9), a large number of observations that were unrelated to the discharge also exhibited significant anomalies. Table 3.10 lists the statistically significant anomalies that were not included in Table 3.8 and Table 3.9 because they could not be ascribed to the discharge of wastewater.

Just as some discharge-related anomalies are not statistically significant, not all statistically significant anomalies are necessarily related to the discharge. Some of these non-discharge anomalies can be identified with particular oceanographic processes that locally alter seawater characteristics, while others occur by chance alone. Even in the absence of effluent discharge, one would expect to find a number of statistically significant anomalies in ambient seawater properties. By the very definition of a 95% confidence level, one '*significant*' departure out of every 20 measurements is expected to occur by chance alone. With the over 2,000 measurements examined for each parameter, it is not surprising that some depart from the mean by slightly more than the 95% confidence interval. In 2000 measurements, approximately 100 significant departures would be expected to occur by chance alone. The 28 anomalies listed in Table 3.10 represent statistically significant departures in the individual half-meter measurements. All have one or more specific characteristics that distinguish them from the discharge-related anomalies identified in

Table 3.8 and Table 3.9. These characteristics are listed in the second column of Table 3.10 and are described below.

Table 3.10 Inventory of Statistically Significant Anomalies that were Unrelated to Effluent Discharge

Survey	Cause	Station	Depth (m)	Property	Magnitude
January	Freshwater Runoff	9	1.0	Salinity	–0.249 ‰
January	Freshwater Runoff	9	1.0	Density	–0.149 σ_t
January	Freshwater Runoff	9	1.0-2.0	Transmissivity	–14.7 %
January	Freshwater Runoff	10	1.0-2.0	Transmissivity	–7.6 %
January	Freshwater Runoff	13	1.5	Salinity	–0.145 ‰
January	Freshwater Runoff	13	1.0-2.5	Transmissivity	–1.2 %
April	BL/Isolated	1	16.5	Transmissivity	–10.2 %
April	BL/Isolated	2	16.0	Transmissivity	–12.7 %
April	Intrusion	7	13.5-14.0	Temperature	–0.19 ºC
April	Intrusion	7	14.5	Density	–0.077 σ_t
April	Intrusion	7	13.5	Dissolved Oxygen	–0.83 mg/L
April	Intrusion/BL/Sign	7	16.0-17.0	Transmissivity	+15.0 %
April	BL/Isolated	12	14.5	Transmissivity	–14.5 %
July	Spike/Isolated/Sign	1	2.5	Salinity	+0.108 ‰
July	Spike/Isolated	2	5.5	Salinity	–0.075 ‰
July	Spike/Isolated	3	3.5	Salinity	–0.096 ‰
July	Spike/Isolated	4	3.5	Salinity	–0.081 ‰
July	Spike/Isolated	7	3	Salinity	–0.072 ‰
July	Spike/Isolated	8	3	Salinity	–0.077 ‰
July	Spike/Isolated/Sign	10	3.5	Salinity	+0.080 ‰
July	Spike/Isolated/Sign	13	1	Salinity	+0.100 ‰
July	Spike/Isolated	14	2.5	Salinity	–0.098 ‰
July	Spike/Isolated/Sign	14	3.0	Salinity	+0.095 ‰
July	Isolated/Sign	13	11.5-17.0	Transmissivity	+6.5 %
October	BL/Isolated	2	16.5-17.0	Transmissivity	–1.5 %
October	Isolated/Sign	6	12.0-15.5	Transmissivity	+3.1 %
October	Equilibration/Isolated	6	14.5-16.0	pH	–0.033 pH
October	BL/Isolated	10	15.5-16.0	Transmissivity	–1.8 %

Freshwater Runoff: Record rainfalls in early January 2005 led to significant runoff of turbid freshwater into the coastal waters of Estero Bay. In the days and weeks prior to the January water-quality survey, a series of powerful winter storms impinged on the Central California coastline dropping more than four inches of rainfall. Substantial runoff carries a high load of terrigenous particulates, which creates patches of turbid tannish-brown freshwater that can be seen near the sea surface along the coastline for several days after a period of heavy rain. Pools of this turbid freshwater were observed visually throughout Estero Bay during the January survey. They were also occasionally encountered during vertical profiling, which led to marked reductions in surface salinity, density, and transmissivity at Stations 9, 10, and 13 (Table 3.10). These turbid pools were clearly unrelated to the effluent discharge because they were restricted to the sea surface and were widely distributed throughout Estero Bay.

Intrusion: Occasionally, observations at the outermost stations are anomalous because of the intrusion of a water mass from outside the survey area. These intrusive features occur as pockets of anomalous properties at the edge of the survey area and are evident in the vertical sections of seawater properties. These anomalies occur because the intrusive features alter the thermocline depth, thereby resulting in a contrast in seawater properties with adjacent, unaffected stations. Such was the case at Station 7 during the April survey. The apparently significant anomalies in temperature, density, and DO that are listed in Table 3.10, were artifacts of the slightly deeper mixed layer that was present at Station 7. The slightly deeper mixed layer meant that water properties within the layer (13.5 m to 14.5 m) were being contrasted with much-different water properties within the shallower thermocline at adjacent stations. It is clear that these anomalies were unrelated to the discharge for two reasons. First, Station 7 is physically separated from the effluent discharge point. Not only does it lie farther offshore than any other station, but anomalies were not apparent at Station 8, which lies between the diffuser structure and Station 7. Secondly, the prevailing flow throughout the April survey was directed toward the south-southwest and not toward Station 7. Instead, the deep lateral anomalies at Station 7 appeared because of slight differences in the depth of the interface between the mixed layer and the deep thermocline.

BL (Boundary Layer): Oscillations in the thickness of the surface or benthic boundary layer, or differences in height of the CTD measurement within that layer, can result in anomalies in water properties that are perceived to be statistically significant. Transmissivity measurements are particularly susceptible to these errors because a sharply defined turbid layer of seawater typically resides immediately above the seafloor. Slight upward displacement of the sharp boundary associated with this turbid benthic layer generates statistically significant differences with the markedly less turbid measurements collected at the same depth level at other stations. A particularly distinct benthic boundary layer was present at some of the stations during the April and October surveys. Variations in the thickness of this layer generated several statistically significant anomalies along its interface (Table 3.10).

Isolation: Most of the anomalies listed in Table 3.10 were isolated. They were not spatially coincident with perceptible anomalies in other water properties, particularly salinity anomalies, which provide the best indicator of the presence of wastewater constituents. Conversely, discharge-related anomalies are often spatially coincident. For example, if there is a perceptible turbidity anomaly (negative transmissivity) associated with the discharge, a negative salinity anomaly will almost always coincide with it. Reduced salinity is the clearest signature of the presence of dilute wastewater constituents. The discharge-related anomalies listed in Table 3.8 and Table 3.9, and shown in the vertical sections, demonstrate that discharge-related perturbations tend to be distinguished by coincident anomalies in a number of independent seawater properties. In contrast, the isolated occurrence of a significant anomaly in only one seawater property probably results from inherent variability in seawater properties.

Spike: Although negative salinity anomalies are often indicative of the presence of wastewater constituents, negative salinity anomalies can also be artificially introduced when the CTD instrument package crosses a sharp thermocline. Salinity is computed from conductivity and temperature probes that do not measure the same water parcel because the sensors are physically separated within the CTD instrument package. In addition, the sensors do not have the same response times. Consequently, when passing though regions of strong vertical temperature gradients, the mismatch between the conductivity and temperature measurements causes erroneous spikes in salinity. Salinity spikes are a common problem with CTD measurements collected within strong vertical thermal gradients. They are routinely observed in MBCSD surveys when the water column is well stratified. Strong thermal stratification in 2005 was present only during the July survey (red line in Figure 3.4 on Page 3-19) and accordingly, it was the only survey that exhibited salinity spiking (Table 3.10). These erroneous salinity spikes were identified from visual inspection of high-resolution vertical profiles of temperature and salinity. Salinity spikes are characterized by marked isolated reductions in salinity within a layer that is less than 0.5 m thick as seen in Figure 3.8de.

Sign: Often, the sign or direction of an anomaly determines its origin. Effluent is less dense, warmer, and much less saline than receiving ocean waters. Negative density and salinity anomalies, combined with positive thermal anomalies, are clear indicators of effluent that is continuing to undergo intense mixing within receiving waters. Conversely, positive transmissivity anomalies reflect less-turbid water, which is not indicative of the presence of wastewater particulates. For example, random variation in the thickness of the boundary layer during the April survey generated a positive transmissivity anomaly in addition to negative anomalies (Table 3.10). Similarly, some of the salinity spikes observed during the July survey reflect increases in salinity. Positive salinity anomalies cannot be generated by a discharge that is comparatively devoid of dissolved solids.

Equilibration: Incomplete equilibration of the pH sensor was apparent in the data collected at the beginning of the October survey at Station 6. The pH sensor requires an extended period of time to fully equilibrate to seawater after exposure to the atmosphere between deployments. While the resulting temporal trend can be reliably removed from the pH measurements, prolonged exposure to the atmosphere between surveys can also affect the initial dynamic range of the pH measurements. During the initial deployment at Station 6 during the October survey, the reduced dynamic range in the pH sensor resulted in pH measurements that were 0.033 lower than the pH measurements taken near the seafloor at surrounding stations. Although small in amplitude, this pH difference was large enough to be statistically significant when compared to the other, nearly uniform pH measurements collected during the survey.

Wastewater-Induced Anomalies

Negative salinity anomalies are usually the only consistent indicator of the presence of wastewater constituents (Table 3.8). Salinity in wastewater is negligible compared to that of the receiving seawater, and a distinct salinity minimum provides *de facto* evidence of the presence of wastewater constituents. All nine of the discharge-related perturbations were associated with salinity anomalies. Five of the nine perturbations exhibited statistically significant reductions in salinity. The remaining four Perturbations (P1, P2, P5, and P6) had salinity anomalies that were visually apparent in the vertical sections (Figure 3.8abde), but were too small to be of significance relative to ambient salinity variability at the time of the surveys.

Salinity is particularly useful as a tracer of wastewater constituents because it directly quantifies the amount of dilution that has been achieved by initial mixing. Specifically, wastewater has essentially no salinity, so C_e can be neglected in Equation 3.1 on Page 3-15, and dilution (D) can be computed from the salinity anomaly ($A = C_o - C_s$) as:

$$D = \frac{-C_o}{\left(C_o - C_s\right)} \equiv \frac{-C_o}{A} \qquad \textbf{Equation 3.2}$$

where:

D = the dilution ratio of the volume of seawater mixed with wastewater,
C_o = the salinity of the wastewater-seawater mixture after dilution by D,
C_s = the background seawater salinity, and
$A = C_o - C_s$ = the salinity anomaly.

The instantaneous dilution factors (*D*) shown in Table 3.8 and Table 3.9 on Page 3-27 were computed using each respective salinity anomaly in Equation 3.2. The computed dilution factors range from 100:1 for the strong salinity anomaly (P9) located within a discharge jet during the October survey, to 807:1 for the weak anomaly (P1) measured during the January survey. Based on the smallest statistically significant salinity anomaly (P4), wastewater-induced salinity anomalies with amplitudes less than approximately |-0.07‰| cannot be reliably discerned against the backdrop of ambient salinity variability. Accordingly, wastewater diluted more than approximately 500 times becomes imperceptible within the variable ocean environment, even with highly sensitive instrumentation capable of resolving extremely small salinity variations (0.0006‰ from Table 3.5 on Page 3-12).

Data collected during the October survey included an opportunistic station (17) that was a first for water-quality monitoring in that it provided high-resolution measurements of wastewater-induced anomalies shortly after discharge. Vertical profiling began at Station 17 in the center of the effluent plume that was visible beneath the sea surface directly over the diffuser structure. As documented by the unusually deep Secchi depths (Table 3.11), seawater clarity was extraordinarily high at the time of the October survey and allowed the effluent plume to be seen within the water column. As shown in Figure 3.8f, the vertical profile at Station 17 passed close (< 1 m) to the diffuser structure and the instrument package measured effluent near the seafloor within a turbulent discharge jet shortly after ejection from a discharge port. The resulting negative salinity and density anomalies had the largest amplitudes ever recorded in two decades of monitoring (Perturbation P9 in Table 3.8 on Page 3-27). This unusual encounter also documented the presence of wastewater particulates as shown by a highly localized reduction in transmissivity (Figure 3.9b).

Table 3.11 Average Secchi Depths (m) measured near the Diffuser and at Distant Locations

Survey	Nearfield[1]	Reference
January	6.1	5.9
April	3.9	3.9
July	3.8	3.8
October	9.0	9.1

[1] Within 30 m of the diffuser structure

It is noteworthy that there were no anomalies in temperature, DO, and pH associated with this perturbation, even though the measurements were made shortly after discharge (Table 3.9 on Page 3-27). This supports the hypothesis that the presence of wastewater particulates does not materially contribute to discharge-related anomalies in temperature, DO, and pH. Instead, as discussed in the following section, observed anomalies in these properties are usually generated by the upward displacement of ambient seawater entrained in the effluent plume. Moreover, the reduction in transmissivity (-3.69%) associated with this perturbation was relatively limited considering the close proximity of the measurement to the point of discharge. It was only apparent because of the consistently high ambient transmissivity (>77%) that prevailed during the October 2005 survey. The high water clarity was also reflected in the unusually large Secchi depths measured during the survey (Table 3.11). For the first time in recent memory, the 18-m euphotic zone reached to the seafloor at all stations.

The presence of wastewater particulates probably also contributed to the increased seafloor turbidity at Station 4 during the July survey (Perturbation P5 in Table 3.8 and Figure 3.9a). However, the high dilution associated with this perturbation (784:1 in Table 3.9), and the fact the prevailing flow would have carried the effluent plume to the north, away from Station 4, suggests that the amplitude of this transmissivity anomaly may have been significantly augmented by natural variations in ambient turbidity. Transmissivity measurements were spatially variable during the July survey and comparatively turbid conditions prevailed overall, as reflected in the shallow Secchi depths listed in Table 3.11. Additionally, all the anomalies in other seawater properties that co-occurred with this anomaly, including the salinity anomaly,

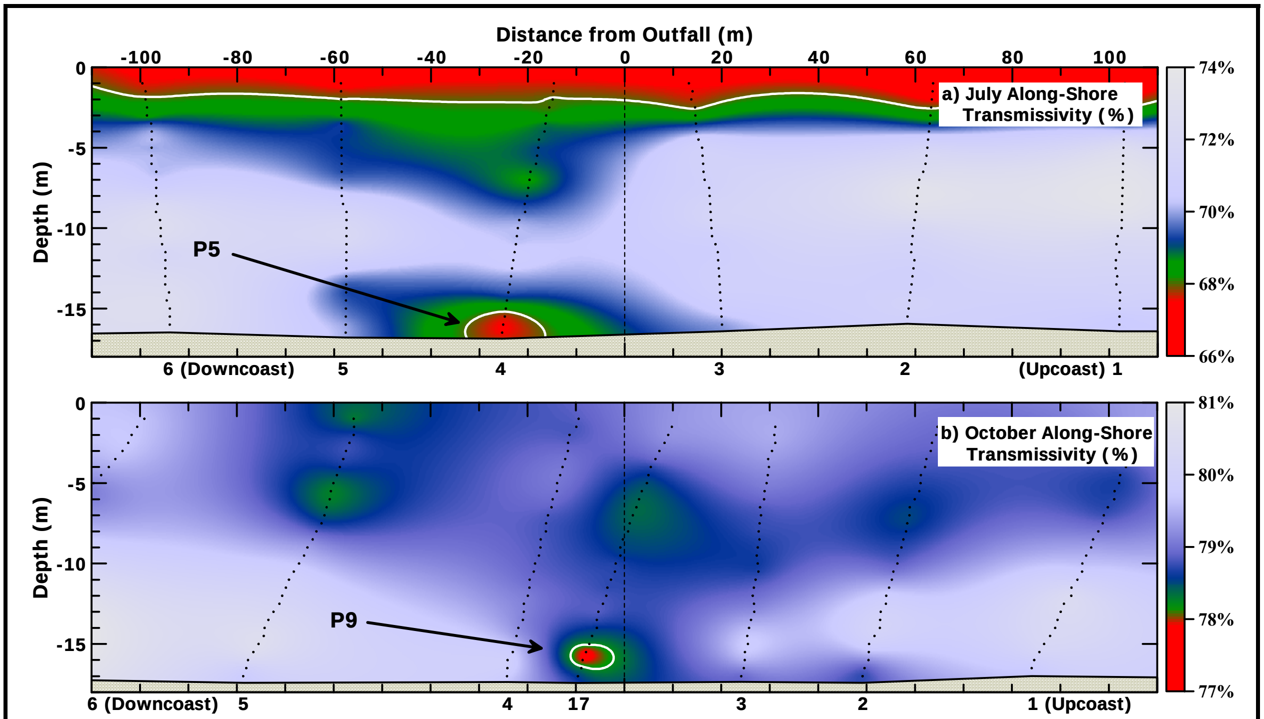

Figure 3.9 Transmissivity Anomalies Induced by the Presence of Wastewater Particulates

were not statistically significant (Perturbation P5 in Table 3.8 and Table 3.9). In fact, the amplitude of the salinity anomaly associated with Perturbation P5 indicates that the wastewater was seven-times more dilute than that of Perturbation P9. Thus, the higher turbidity associated with Perturbation P5 is incongruous with wastewater particulate loading.

The very large salinity anomaly associated with Perturbation P9 indicates that turbulence caused by the momentum of the jet rapidly dilutes the wastewater by more than 100-fold immediately upon ejection from a discharge port. The measurement was made 2 m above the seafloor within the turbulent discharge jet emanating from one of the 28 diffuser ports. These very nearfield measurements demonstrate that mixing within the turbulent discharge jet achieves dilutions comparable to the final dilution (133:1) predicted by conservative modeling after a 9 m rise of the plume through the water column. Thus, the momentum of the discharge jet alone is capable of achieving dilution levels close to the permit-specified dilution ratio, without even considering the additional dilution achieved during the plume's rise to its equilibrium depth within the water column. The extremely low density measured within the discharge jet, as reflected by an anomaly of –0.274 σ_t in Table 3.8, indicates that the plume was highly buoyant, and would rise rapidly through the water column as it mixed with surrounding seawater. Accordingly, a dilution of 346-fold was achieved by the time the effluent plume rose an additional 11.5 m in the water column (Perturbation P8 in Table 3.8). At that point, 4 m below the sea surface, the turbidity associated with wastewater particulates was almost imperceptible despite the high water clarity that prevailed at the time (Figure 3.9b).

This explains why the discharge consistently meets receiving water limitations, and why the presence of dilute wastewater particulates is rarely detected in the upper water column beyond the ZID. Specifically, the salinity measurements collected close to a diffuser port demonstrate that wastewater is diluted by more than 100-fold by the momentum of the discharge jet alone. Subsequent buoyancy-induced mixing further dilutes the wastewater to levels at least 2.5-times the 133:1 critical initial dilution used to establish permit limitations on contaminant concentrations within the wastewater discharged from the MBCSD treatment plant. These dilution computations demonstrate that, during 2005, the outfall was operating far more efficiently than predicted by modeling, and was rapidly diluting effluent more than 340-fold within the ZID. Consequently, the COP receiving-water objectives were being easily met by the discharge. Based on dilution ratios measured in 2005, contaminant concentrations within wastewater could have been 2.5 times higher than the limits specified in the current NPDES permit, and still would have met the COP guidelines.

Because of the high dilution achieved shortly after wastewater discharge, it is not surprising that there is little evidence of the presence of wastewater constituents in the measurements of seawater properties ther than salinity and density. Instead, discharge-related anomalies in the three water-quality parameters (transmissivity, DO, and pH) are largely generated by entrainment of ambient seawater within the rising effluent plume. These entrainment-generated anomalies are discussed in the following section. The only exceptions were the two transmissivity anomalies that were observed close to the seafloor in Perturbations P5 and P9. In addition, negative anomalies in two of the physical properties, namely, salinity and density, could only have been induced by the presence of dilute wastewater constituents. However, the negative density anomalies measured in six of the nine perturbations indicate that the plume was buoyant within those perturbations, and that additional initial mixing would be expected to occur as the plume continued to rise. Consequently, the dilution estimates reported in Table 3.8 for Perturbations P2 through P6, and P8, underestimate the level of initial dilution that was ultimately achieved by the discharge. Initial dilution is considered complete when the plume reaches buoyant equilibrium.

Entrainment-Generated Anomalies

In contrast to the wastewater-induced anomalies described above, all other anomalies in water-quality properties listed in Table 3.9 were generated by the upward displacement of ambient seawater that was entrained in the buoyant effluent plume. This was also the case for all of the thermal anomalies listed in Table 3.8. This type of discharge-related anomaly is generated when ambient bottom water is entrained in the rising effluent plume. After being displaced upward, bottom-water properties are juxtaposed with shallow-water properties, and the contrast becomes apparent as an anomaly. The anomalies arise because of a difference in the properties of the ambient water near the seafloor, and the seawater properties in the upper water column. For example, seawater immediately above the seafloor tends to be naturally low in pH and DO. Also, increased particulate loads from sediment resuspension can cause seafloor transmissivity to be reduced. Except for the transmissivity anomalies in Perturbations P5 and P9, all of the negative anomalies found in pH, DO, transmissivity, and temperature can be linked to ambient conditions near the seafloor.

Perturbation P1 from the January survey provides a striking visual example of how entrainment generates anomalies. The five vertical sections shown in Figure 3.10 clearly reveal an upward extension of seafloor properties within a localized region around Station 3. Intrusion of a deep watermass during the January-2005 survey caused seawater immediately above the seafloor to be naturally low in temperature, pH, and

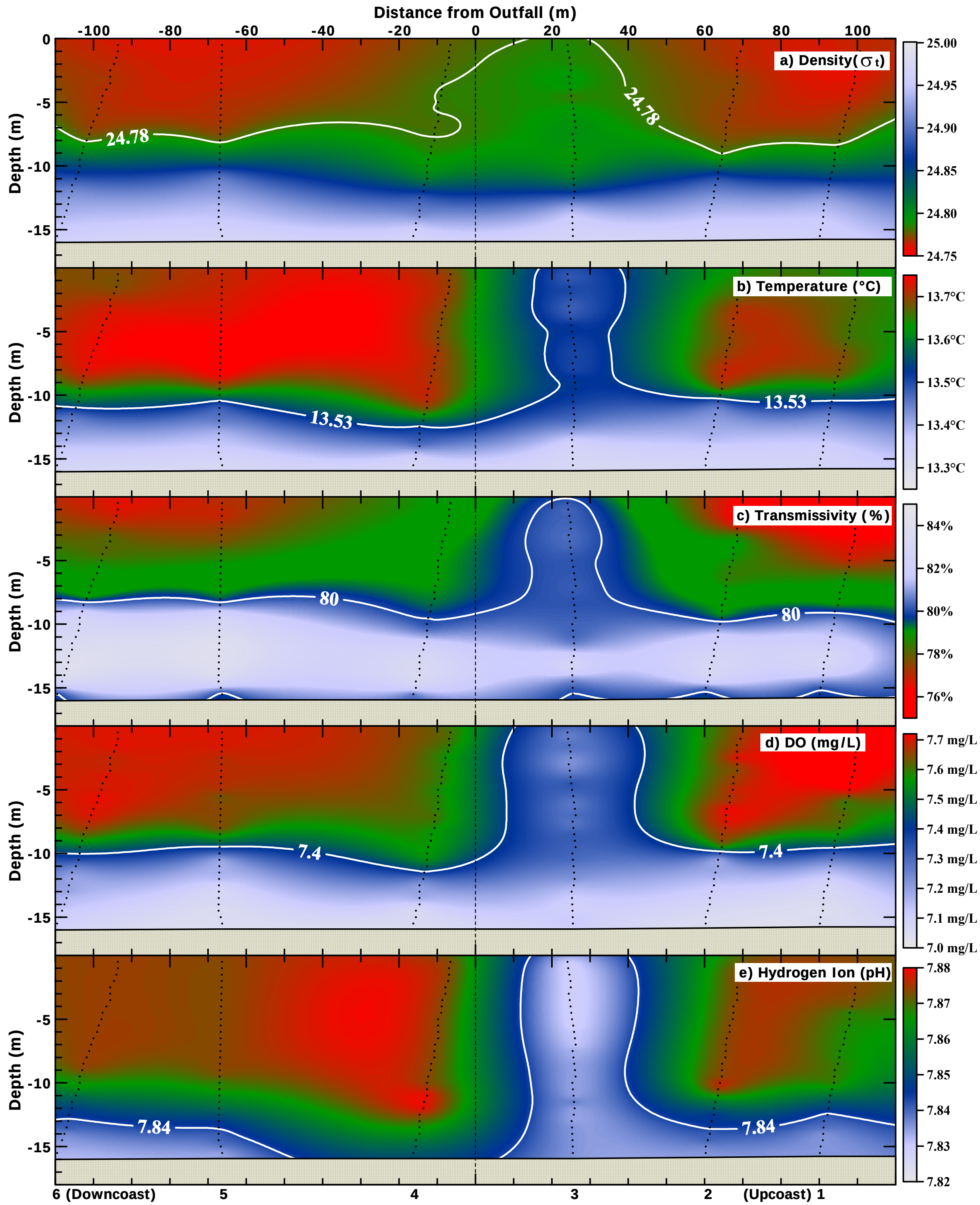

Figure 3.10 January Along-Shore Distribution of Seawater Properties Characterizing Perturbation P1

DO, and high in density and transmissivity. Figure 3.10 displays the unmistakable connection between these unusual seawater properties at depth, and the properties measured near the sea surface at Station 3.

Entrainment-generated anomalies only become apparent when ambient seawater properties near the seafloor are distinctly different from those of the rest of the water column, as is the case when receiving waters are vertically stratified. Without distinctive properties at depth, entrainment of deep ambient seawater within the rising effluent plume would not produce obvious differences between the entrained water and shallow water properties. Thus, the entrained bottom seawater acts as a tracer of the effluent plume after discharge. These entrainment-generated anomalies could have been produced by the discharge of warm seawater, containing no suspended solids or other contaminants whatsoever. Conversely, none of the anomalies would have been generated by the discharge if ambient receiving waters had lacked vertical stratification.

Not only is the connection between the shallow entrainment-generated anomalies and the ambient seafloor properties clear from Figure 3.10, but the character of some of the anomalies demonstrates why they could not have been induced by the presence of wastewater constituents. Wastewater is less dense, warmer, and carries higher suspended particulate loads than receiving ocean waters. If wastewater properties were materially contributing to the anomalies, then positive thermal and negative transmissivity anomalies would result. Instead, the thermal anomaly was negative, indicating that the perturbation was cooler than surrounding waters, a phenomenon that could only be produced by the entrainment of cooler bottom water, and not the presence of warmer wastewater.

The positive transmissivity anomaly is even more diagnostic. Figure 3.10c shows that low-turbidity seawater above the seafloor, shown in blue, was moved upward into the water column by the buoyant effluent plume. As a result of the upward movement of this low-turbidity seawater toward the sea surface, the transmission of natural light was actually increased. This is opposite of the negative impact on ambient light transmission that is normally thought to be produced by discharge of wastewater particulates. It emphasizes why many of the receiving-water limitations specified in the COP explicitly state that limitations only apply to impacts caused by the presence of wastewater constituents, and, by implication, not to changes generated by the entrainment of ambient seawater. This is also implied by the 95% significance tests, which restrict attention to discharge-induced changes beyond the range in ambient seawater properties. In any regard, entrainment-generated impacts, beneficial or otherwise, are largely dictated by the existing ambient seawater stratification rather than the quality of discharged wastewater.

3.3.3 Compliance with Receiving-Water Limitations

As in prior years, measurements collected during the year-2005 surveys demonstrated that the discharge fully complied with receiving water limitations. Irrespective of the cause of the discharge-related perturbations in seawater properties, none of the perturbations constituted an exception to the limitations promulgated by the NPDES discharge permit, the COP, or the central-coast Basin Plan. Even within the ZID, where exceptions to numerical limitations in the COP are allowed, most of the observed discharge-related anomalies were too small to be reliably detected even at the 95% confidence level. In general, only slight salinity anomalies and a couple of transmissivity observations revealed the presence of very well mixed effluent. Those observations demonstrated that discharged wastewater was undergoing rapid dilution immediately after discharge. Overall, water-quality measurements were indicative of a diffuser structure that was operating at a level better than originally designed.

Each of the discharge-related anomalies listed in Table 3.8 and Table 3.9 was evaluated for compliance with the receiving-water limitations. Specifically, the limits stated in the NPDES discharge permit were not exceeded because the discharge did not cause:

1. *the temperature of the receiving water to adversely affect beneficial uses;*
2. *a significant reduction in the transmittance of natural light at any point outside the initial dilution zone;*
3. *the pH outside the zone of initial dilution to be depressed below 7.0, raised above 8.3, or changed more than 0.2 units from that which occurs naturally; or*
4. *the dissolved oxygen concentration outside the zone of initial dilution to fall below 5.0 mg/L or to be depressed more than 10 percent from that which occurs naturally.*

Other than density and salinity anomalies, the only statistically significant discharge-related anomalies found beyond the ZID during the 2005 surveys were the pH anomaly associated with Perturbation P3 and the transmissivity anomaly associated with Perturbation P5. Discharge limitations are not imposed on the salinity and density fields. Compliance with each of the four receiving-water limitations listed above is addressed below.

Temperature

The NPDES discharge permit states that wastewater-induced temperature changes should not adversely affect beneficial uses. This requirement was based on the policies promulgated by the California Thermal Plan (SWRCB 1972). Beneficial uses designated for the waters within northern Estero Bay were discussed previously in Section 2.1.2. Beneficial uses that could conceivably be affected by large fluctuations in seawater temperature include recreation, marine habitat, shellfish harvesting, and fishing. However, the amplitudes of all eight discharge-related thermal anomalies identified in Table 3.8 were less than 0.5°C. These thermal anomalies would not tangibly affect beneficial uses such as water-contact sports or marine habitat because they are small in amplitude, are highly localized within a restricted depth range, and occur in an area well offshore where few water-contact activities take place. The discharge-related reductions in near-surface temperature are particularly negligible when compared to the 5°C range in ambient sea surface temperatures that typically prevail along the central coast at any given time. Compare, for example, the range in sea-surface temperatures shown in the satellite image on the cover of this report.

Furthermore, only one of the thermal anomalies (P7) was found to be statistically significant compared to natural fluctuations in the ambient ocean environment that prevailed during each respective survey. However, as discussed above, this reduction in temperature was due to the upward displacement of ambient bottom water and not the presence of effluent itself. The presence of warm effluent would cause a positive thermal anomaly, the exact opposite of all eight of the observed anomalies. Because it was generated by the upward displacement of cool bottom seawater, and not by the presence of warm wastewater constituents, it represents a temperature variation that was naturally present in the marine environment at the time of the discharge.

Water Clarity

The NPDES waste discharge requirements place restrictions on impacts to water clarity resulting from the discharge. Specifically, they dictate that the discharge shall not cause a '…*Significant reduction in the transmittance of natural light at any point outside the initial dilution zone*.' This requirement derives from COP Objective C.3 in Table 3.1. There were no exceptions to this requirement in the water-quality data collected during 2005, even though significant discharge-related decreases in transmissivity (water clarity) were detected in Perturbations P5 and P9 (Table 3.9).

The discharge-related decrease in transmissivity associated with Perturbation P9 during the October survey constituted a relatively rare measurement of effluent particulates as they were being dispersed within the ZID (Figure 3.9b). Because the significant decrease in transmissivity occurred within the ZID, the limitation that pertains to the penetration of natural light does not apply.

The discharge-related decrease in transmissivity associated with Perturbation P5 during the July survey was measured beyond the ZID but, as discussed previously, it is unclear how much the presence of wastewater particulates contributed to the turbidity at that point. In any regard, the significant reduction in transmissivity only extended 1.5 m above the seafloor. Because the significant reduction in water clarity was restricted to the seafloor, it does not constitute an exception to the COP objectives. Because it was located just above the seafloor, it could not have caused "*Natural light to be significantly reduced….*" Based on the deepest measured Secchi depths of 3.8 m (Table 3.11), the bottom of the euphotic zone was approximately 8.0 m, indicating that very little natural light was reaching the seafloor where the statistically significant anomalies in transmissivity were observed.

Insofar as the transmission of natural light measured by Secchi depths in Table 3.11, there were no significant reductions in average Secchi depth observed near the diffuser during the 2005 surveys. In fact, during the January survey, nearfield Secchi depths extended to a slightly greater depth than distant stations. This occurred because the Secchi depths measured near the outfall happened to fall outside the turbid pools of freshwater runoff that were distributed along the coastline at the time of the survey.

Alkalinity (pH)

The NPDES permit places the following restriction on discharge-related changes to the pH of receiving waters: '*The discharge shall not cause:…pH outside the zone of initial dilution to be depressed below 7.0, raised above 8.3, or changed more than 0.2 units from that which occurs naturally.*' Compliance with these receiving water limitations is easily demonstrated because the wastewater itself had a pH that remained between 7.3 and 7.9 throughout 2005 (see Chapter 2). Moreover, the difference between the average pH of discharged wastewater (7.58) and receiving waters (7.65) was negligible. Because of the well-buffered nature of the carbonate system in seawater, the slightly increased acidity of the wastewater, which was on average only 0.06 pH units lower than that of receiving waters, would be quickly and easily assimilated shortly after discharge.

As expected from the similarity between the pH of seawater and wastewater, all of the pH anomalies attributable to the discharge were generated by entrainment processes rather than the presence of dilute wastewater. All four of the slight, but statistically significant pH anomalies (Perturbations P1, P3, P7, and P8 in Table 3.9 on Page 3-27) were associated with the upward displacement of ambient seawater, which is naturally low in pH at depth. Seawater near the seafloor is slightly more acidic due to the production of carbonic acid (dissolved carbon dioxide CO_2) by biotic respiration and decomposition. The naturally occurring vertical difference in pH was evident during the January survey as shown in Figure 3.10e. As a result, entrainment of low pH bottom water within the rising effluent plume generated a statistically significant near-surface pH anomaly (Perturbation P1). The entrainment-generated pH anomaly at Station 3, and its connection to the seafloor pH, is visually apparent in the vertical section. However, as with the other perturbations, the amplitude of this pH anomaly was less than 0.05 standard pH units, consequently, none of the anomalies can be considered changed by '*…more than 0.2 pH units from that which occurs naturally*.'

Dissolved Oxygen

As with the other water-quality parameters, all receiving-water measurements of dissolved oxygen (DO) that were collected during 2005 complied with the requirements of the NPDES discharge permit. The receiving water limitations in the NPDES permit state that the '*…discharge shall not cause…dissolved oxygen concentration outside the zone of initial dilution to fall below 5.0 mg/l or to be depressed more than 10 percent from that which occurs naturally*.' The first requirement derives from the Central Coast's Basin Plan (RWQCB 1994) which states that, in addition of the provisions of the COP, '*The mean annual dissolved oxygen concentration shall not be less than 7.0 mg/l, nor shall the minimum dissolved oxygen concentration be reduced below 5.0 mg/l at any time*.'

The second limitation derives from COP Objective D-1 in Table 3.1. It qualifies the cause of the oxygen depression to be the '*…result of the discharge of oxygen demanding waste materials*.' As described in Section 3.2.4 on Page 3-18, a reduction in DO within receiving waters can be a concern with wastewater discharge into enclosed water bodies. In some cases, the seafloor accumulation of oxygen-demanding materials within the wastewater particulates can deplete receiving water of its oxygen content and impact marine and estuarine organisms. Oxygen depletion, however, is rarely a problem with open-ocean discharges. Also, as described in Section 2.2.3, the MBCSD treatment process consistently removes most of the oxygen-demanding materials from the influent stream. As a result, the plant wastewater's biochemical oxygen demand (BOD_5) has been far below the permit maximum; not only in 2005, but throughout the past two decades of operation (See Section 5.1). In addition to its low BOD_5, the MBCSD discharge occurs in a particularly well-flushed open-ocean environment where wastewater particulates are rapidly dispersed shortly after discharge.

During 2005, all of the discharge-related reductions in DO were generated by the upward displacement of ambient seawater that was naturally depleted in oxygen. Because the DO limitation only applies to reductions that occur "*…as a result of the discharge of oxygen demanding waste materials*," it does not apply to reductions in DO caused by the movement of ambient waters even though they are generated by the physics of the discharge.

Furthermore, none of the DO anomalies were subject to the limitations because they were either not statistically significant (Perturbation P1 through P6), or were recorded well within the ZID (Perturbations P7 and P8). Even so, all of the DO anomalies complied with the numerical limits specified in the permit. Specifically, none of the anomalous DO concentrations fell below the 5 mg/L minimum specified in the Basin Plan and the NPDES discharge permit. Interestingly, this contrasts with some measurements of ambient DO concentrations collected during the April and October surveys. Both surveys had DO measurements that dropped below 5 mg/L at many stations within a benthic water mass that was naturally depleted in oxygen due to its deep offshore origin. As described in Section 3.2.4, the shoreward movement of a deep, oxygen-poor water mass is characteristic of upwelling conditions. The deep water mass originates in the Southern California Bight and is carried northward below the sea surface by the Davidson Undercurrent. Near the diffuser, this water mass can be entrained in, or pushed ahead of the rising effluent plume, resulting in shallow discharge-induced DO anomalies. The DO depletion in the seafloor water mass occurs naturally because it has not been in recent contact with the atmosphere, and ambient respiration and decomposition processes have slowly removed oxygen from the seawater. Given the presence of these naturally low DO concentrations, none of the discharge-related anomalies can be considered "*…to be depressed more than 10 percent from that which occurs naturally*."

3.3.4 Coliform Bacteria

The surfzone along Atascadero State Beach was monitored to assess bacteriological levels, aesthetic conditions, and compliance with the NPDES permit and the COP. MBCSD WWTP personnel monitored the surfzone stations on a weekly basis during the summer months (May through October) and at least monthly during the winter months (November through April). Samples were analyzed for total and fecal coliform densities. A regional map depicting the eight surf-zone stations is presented in Figure 3.1 on Page 3-5 and the coordinates of the stations are listed in Table 3.2 on Page 3-4. During 2005, as in prior years, the MBCSD discharge was in compliance with the surfzone bacterial objectives specified in the NPDES permit. Although there were localized isolated instances of elevated beach coliform, none were related to the discharge of wastewater by the treatment plant. This is not surprising because the effluent is disinfected prior to discharge.

The small creeks feeding into northern Estero Bay have long been identified as the source of elevated coliform contamination of the surfzone along Atascadero State Beach. This non-point source contamination is unrelated to wastewater discharge through the offshore outfall. The mouth of Morro Creek lies close to the treatment plant, and, in 1993, Station G in the creek's outflow was added to the surfzone monitoring program to document the time-varying influence of non-point-source coliform input from nearby creeks. Station G is sampled only when Morro Creek is flowing into the ocean, and its coliform densities are not used to assess numerical compliance with the bacteriological objectives in the NPDES permit. Instead, coliform densities in the creek's outflow aid in the interpretation of surfzone bacteriological measurements by providing an index of non-point source contamination in samples collected at the regularly sampled surfzone stations (Stations A1 through F).

During 2005, Morro Creek was lagooned for a relatively short period between the second week in September and the last week in October (Table 3.12 on Page 3-45). It became lagooned again sometime after the first week in November. As a result, samples were collected at Station G on 26 of the 33 surfzone sampling occasions during 2005. All but one of these 26 Morro Creek samples were contaminated with total and fecal coliform densities at or above 130 MPN/100 ml. Ten of these samples were heavily contaminated with total coliform densities at or above 1600 MPN/100 ml. Fecal coliform densities within these samples were correspondingly high, suggesting that most of the bacterial contamination in the creek was of fecal rather than vegetative origin. Large herds of grazing cattle within the watershed are probably largely responsible for the creek contamination.

Despite significant bacterial input from creek outflow and other non-point sources in the region, coliform organisms were not detected in over a third (39%) of the 225 surfzone samples. When coliform organisms were detected, they were usually found at densities near the detection limit of 2 MPN/100 ml. Approximately 59% of the samples had coliform densities at or below 2 MPN/100 ml.

Most of the shoreline samples collected at the regular surfzone monitoring stations during 2005 contained coliform densities that were below the most stringent bacteriological standard specified in the NPDES discharge permit. Only ten samples had total coliform densities exceeding 70 MPN/100 ml. These elevated measurements are highlighted in bold in Table 3.12. For the reasons described below, all were demonstrably unrelated to effluent discharge.

First, there was no correlation between surfzone samples with elevated coliform densities and the offshore discharge of wastewater by the treatment plant. As shown by the bolded values in the "*effluent*" column of Table 3.12, only four effluent samples had total coliform densities that exceeded 70 MPN/100 ml on the days that surfzone samples were collected. However, none of these occasions coincided with measurements of elevated coliform densities in any of the surfzone samples. On each of the seven days that elevated total coliform densities were measured in surfzone samples, the corresponding effluent coliform density was less than one-sixth of the maximum density measured in the surfzone samples. Considering that wastewater is diluted by at least two orders of magnitude immediately after discharge, it is inconceivable that coliform densities measured in the effluent could materially contribute to the elevated surfzone densities.

Second, three of the ten elevated total coliform densities measured in the surfzone samples were not associated with elevated fecal coliform densities (Stations C and E on 5 January, and Station E on 10 January). Two additional samples (Stations E on 11 April and Station E on 9 May) had fecal coliform densities that were well below total coliform densities. Discharged wastewater would be expected to contribute only fecal coliform. Instead, the elevated total coliform densities measured in all five samples arose largely from vegetative rather than animal sources. Decay of kelp and other plant material along the beach was probably largely responsible for the elevated non-fecal coliform densities observed in these surfzone samples.

Third, half of the elevated coliform densities were measured in surfzone samples collected at Station E, which were directly impacted by the discharge of coliform-contaminated water from Morro Creek. Normally, the mouth of Morro Creek lies well downcoast of Station E, as shown in Figure 3.1 on Page 3-5. However, in 2004, the mouth of Morro Creek migrated northward to a location directly in front of the treatment plant where it remained throughout 2005 (Figure 3.11). As a result, seawater samples collected within the surfzone at Station E incorporated a substantial fraction of Creek outflow. As discussed previously, Morro Creek is one of several creeks feeding into Estero Bay that have been recognized as sources of elevated coliform contamination of the surfzone along Atascadero State Beach. Normally, creek-water samples collected at Station G, which is within the Morro Creek outflow, are intended to be an index of this non-point source contamination within surfzone waters as a whole. However, in 2005, because Station G was directly onshore of Station E, its samples were particularly indicative of creek contamination at Station E.

Fourth, the two elevated coliform densities measured in samples collected on 2 November were from reference stations (A1 and F in Table 3.2 on Page 3-4). These stations were selected as control stations because they were considered too distant from the offshore discharge to be influenced by it.

Fifth, five of the elevated coliform measurements (Stations C and E on 5 January, Station E on 10 January, Station E on 11 April, and Station E on 9 May) coincided with rainfall events (shown in the rightmost column of Table 3.12). Runoff during these events washes non-point source material into the ocean.

In particular, fecal material, such as pet wastes, that accumulate onshore during dry spells are discharged into the ocean over a relatively short period during rainstorms.

Sixth, two of the elevated coliform densities (Station E on 10 January and Station F on 2 November) were measured in samples collected near identifiable debris sources near the surfzone.

Figure 3.11 Aerial View of the Morro Creek Outflow Location

Finally, isolated samples containing densities exceeding 70 MPN/100 ml still meet the requirement for areas where shellfish is harvested for human consumption (see Standard A.2 in Table 3.1 on Page 3-3) because the limit applies to a thirty-day median rather than to isolated measurements. Six of the ten elevated coliform measurements were the only elevated measurements recorded at a given station within any individual month. Station E was the only station where elevated coliform densities were repeatedly measured. As described above, the elevated coliform measurements recorded at Station E in the early part of 2005 resulted from Morro Creek outflow and other non-point source contamination associated with rainfall runoff.

As in prior years, none of the elevated coliform densities measured in the surfzone samples collected in 2005 could be ascribed to the MBCSD wastewater discharge. The wastewater is disinfected prior to discharge and consequently, has a consistently low coliform density. With the 100-fold dilution achieved within a few meters of the discharge, discharge-related coliform densities in the receiving water beyond the ZID are imperceptible. Furthermore, the discharge occurs well offshore and prevailing currents are rarely directed toward the shoreline. In contrast, onshore non-point sources are responsible for contamination events observed in the surfzone, and their influence heavily confounds interpretation of any potential contribution from the effluent discharge.

Table 3.12 Effluent and Surfzone Coliform Bacteria (MPN/100 ml), and Rainfall (inches) during 2005

		Total Coliform Bacteria									Fecal Coliform Bacteria								
Date	Effluent	A1	A	B	C	D	E	F	G	A1	A	B	C	D	E	F	G	Rainfall [1]
05-Jan	13	27	22	4	**170**	17	**≥1600**	50	**1600**	9	2	<2	23	4	50	30	**240**	0.07
10-Jan	23	—	—	—	—	—	**220**	—	—	—	—	—	—	—	70	—	—	0.90
07-Feb	2	2	<2	<2	<2	<2	4	9	**900**	2	<2	<2	<2	<2	2	2	**900**	—
14-Mar	<2	23	4	15	17	**80**	**80**	8	**≥1600**	23	4	4	8	**80**	**80**	4	**300**	—
11-Apr	<2	6	13	2	2	8	**500**	17	**900**	6	13	2	2	8	**170**	17	**240**	0.11
02-May	2	7	<2	2	13	8	14	4	**240**	4	<2	2	4	8	11	2	**130**	0.30
09-May	23	11	27	2	4	23	**140**	4	**1600**	4	27	<2	4	8	**90**	4	**1600**	0.90
16-May	<2	<2	<2	2	<2	<2	4	<2	**500**	<2	<2	2	<2	<2	2	<2	**500**	—
23-May	<2	2	2	<2	<2	<2	<2	<2	**500**	2	2	<2	<2	<2	<2	<2	**500**	—
31-May	<2	<2	<2	<2	<2	<2	4	17	**1600**	<2	<2	<2	<2	<2	4	17	**1600**	—
06-Jun	<2	<2	<2	<2	<2	2	<2	<2	60	<2	<2	<2	<2	2	<2	<2	60	—
13-Jun	<2	<2	<2	7	<2	2	4	<2	**240**	<2	<2	7	<2	2	2	<2	**240**	0.03
20-Jun	7	4	2	<2	4	2	<2	2	**500**	2	2	<2	2	2	<2	2	**500**	0.03
27-Jun	4	<2	22	4	<2	2	11	<2	**300**	<2	22	4	<2	2	11	<2	**300**	—
05-Jul	<2	4	4	2	4	2	<2	2	**500**	4	4	2	4	2	<2	2	**500**	—
11-Jul	<2	2	4	<2	2	**300**	4	<2	**130**	2	4	<2	<2	**300**	4	<2	**130**	—
18-Jul	<2	<2	<2	<2	<2	2	<2	<2	**300**	<2	<2	<2	<2	2	<2	<2	**240**	—
25-Jul	26	<2	<2	23	17	23	13	17	**≥1600**	<2	<2	23	17	23	13	17	**≥1600**	—
01-Aug	17	<2	<2	2	2	2	14	2	**500**	<2	<2	2	2	2	14	2	**500**	—
08-Aug	**130**	<2	4	4	2	<2	2	2	**1600**	<2	4	4	2	<2	<2	2	**800**	—
15-Aug	<2	<2	<2	2	4	4	14	23	**≥1600**	<2	<2	2	4	4	14	23	**240**	—
22-Aug	<2	22	13	<2	2	2	4	13	**300**	22	13	<2	2	<2	4	13	**300**	—
29-Aug	2	2	2	<2	<2	<2	2	<2	**300**	2	2	<2	<2	<2	2	<2	**300**	—
06-Sep	50	2	17	<2	<2	4	<2	<2	**≥1600**	2	17	<2	<2	4	<2	<2	**≥1600**	—
12-Sep	4	<2	13	2	<2	<2	11	<2	**300**	<2	13	2	<2	<2	11	<2	**240**	—
20-Sep	<2	11	4	50	4	2	2	13	—	11	2	50	4	2	2	13	—	—
26-Sep	<2	<2	4	<2	<2	4	<2	11	—	<2	4	<2	<2	4	<2	11	—	—
03-Oct	**240**	2	<2	<2	2	2	4	<2	—	2	<2	<2	2	2	4	<2	—	—
10-Oct	**110**	8	<2	23	<2	<2	<2	2	—	8	<2	4	<2	<2	<2	2	—	—
18-Oct	2	7	<2	<2	2	<2	2	4	—	4	<2	<2	2	<2	2	4	—	—
25-Oct	2	2	2	<2	<2	4	11	4	**≥1600**	2	2	<2	<2	4	11	2	**≥1600**	—
02-Nov	26	**500**	13	8	2	<2	13	**130**	**≥1600**	**500**	8	8	2	<2	8	**130**	**≥1600**	—
05-Dec	**300**	<2	<2	<2	11	<2	<2	4	—	<2	<2	<2	11	<2	<2	4	—	0.50

[1] Cumulative rainfall in inches on the day of surfzone sampling and over the four days prior to sampling.

CHAPTER 4

Marine Sediments and Benthic Biota

4.0 MARINE SEDIMENTS AND BENTHIC BIOTA

This chapter describes the coastal benthos within Estero Bay where the MBCSD effluent is discharged. The MBCSD Offshore Monitoring Program includes benthic observations to assess potential impacts of wastewater discharge on marine sediments near the outfall, and on infaunal communities that live within them. Benthic environments are important indicators of the presence of marine pollution. They are the major reservoir and sink for most contaminants that enter the ocean. Deleterious changes in the benthic environment, resulting from the deposition of effluent particulates, have been observed around other coastal outfalls (e.g., Stull et al. 1986a). However, no such impacts have been found after two decades of benthic monitoring around the MBCSD outfall. This includes the October 2005 survey described in this chapter.

Both the physicochemical and biological properties of the benthos are described in this chapter. Dissolved, suspended, and sinking types of organic and inorganic matter are dynamically interrelated in the marine ecosystem. The quality of this material directly affects the health of infaunal communities. Infauna residing within sedimentary environments serve as indicators of contaminated sediments because of their limited mobility and well-defined responses to pollution. Numbers of species, abundance, biomass, and other parameters of infaunal community composition can indicate contaminant-caused stresses if, for example, gradients extending from a pollutant source to distant unaffected areas are observed.

This chapter presents results from a benthic survey conducted in October 2005. After 1998, semiannual surveys were eliminated because seasonal variability had been well determined by winter and summer surveys conducted over the previous 13 years. In lieu of two surveys each year, the single October survey increased spatial resolution around the outfall by adding four stations at nearfield distances from the diffuser structure. The decrease in sampling frequency and increase in spatial resolution was incorporated in the revised Benthic Monitoring Program (RWQCB-USEPA 1998b) after a collaborative review of historical data by MBCSD consultants and the staff of the MBCSD, RWQCB, and USEPA (USEPA 1998; RWQCB-USEPA 1998c; MBCSD 1997b, 1998; MRS 1997ab, 1998efgh).

The reporting schedule was also changed so that separate benthic monitoring reports were no longer generated. Instead, the full results of the October 2005 benthic survey are presented in this annual report. Physicochemical data from the survey are tabulated in Appendix D of this report, and benthic biological data are provided in Appendix E. BC Laboratories conducted the chemical analyses following methods described by Tetra Tech (1986ab). Marine Research Specialists performed the infaunal community analyses and data synthesis.

The following section describes the benthic-monitoring program, including collection methods and station design. The second section of this chapter presents the findings related to the physicochemical analysis of the sediments. The last section describes the health of the benthic infaunal community residing within Estero Bay during 2005.

4.1 MONITORING PROGRAM

As with water-quality monitoring described in Chapter 3, sediment samples were collected at reference and outfall stations to investigate any potential discharge-related impacts to the benthos. The samples were analyzed for physicochemical properties and infaunal community composition in order to compare the benthic environment at these two sets of stations, and at gradient stations in between.

Figure 4.1 Benthic Sampling Stations

4.1.1 Scope of Monitoring

On 4 and 5 October 2005, offshore sediment samples were collected at nine stations within Estero Bay (Figure 4.1). The field sampling and laboratory procedures adhered to the requirements of the current Monitoring and Reporting Program (Number 98-15; RWQCB-USEPA 1998ab). This was the seventh benthic survey to implement the changes to the benthic-monitoring program specified in the NPDES discharge permit. During the two decades of offshore monitoring, the benthic-monitoring program has undergone a number of major modifications. The original permit, issued in 1985, specified semiannual sediment collection at six stations with physicochemical analysis of a composite of three replicate grab samples collected at each station. The stations were centered around the outfall along a north-south isobath. Five of these original stations have continued to be sampled throughout the monitoring program. The sixth station, a '*reference*' station located 1 km to the south, was dropped when the permit was reissued in 1993 because it was unrepresentative of the benthic environment near the outfall.

The discharge permit issued in 1993 (RWQCB-USEPA 1993ab) also changed laboratory protocols. It initially required separate chemical analysis of each replicate sample collected at the original five stations. Separate analyses of replicate samples quantified the within-station variability and lent insight into the statistical significance of spatial differences among stations. In particular, this report uses the replicate chemical analyses conducted in 1993 and 1994 to establish confidence intervals around concentrations measured at each station. In 1995, chemical analysis reverted to a composite protocol where aliquots from each of the three replicates collected from a station were homogenized into a single composite sample in the laboratory before extraction. Other modifications included changing the frequency of analysis for '*remaining priority pollutants*' and '*other pesticides*' from a semiannual basis to an analysis of only the samples collected during the October survey.

Based on an analysis of 13 years of benthic data collected as part of the MBCSD monitoring program, further modifications were implemented when the current NPDES permit was issued in 1998 (RWQCB-EPA 1998ab). These changes included the elimination of the post-winter survey, and the addition of four benthic sampling stations close to the outfall. After 13 years, the amplitude of seasonal fluctuations had been adequately determined. As a result, the focus of monitoring shifted to the determination of interannual changes in spatial patterns. To that end, four additional stations, located 60 m from the outfall, were added to the five historical stations. The cross-shore configuration of two of these new stations contrasts with the purely along-isobath pattern historically sampled. These cross-shore stations were added to provide a more spatially balanced sampling pattern capable of detecting potential impacts that may not be restricted to along-shore directions. Finally, replicate chemistry samples were no longer required in the current permit. Instead, chemical analyses were performed on a single grab sample collected at each of the nine stations.

4.1.2 Sampling Station Design

Regular benthic sampling was conducted at the nine stations shown in Figure 4.1 on Page 4-2 and in Figure 4.2 on Page 4-7. The target positions for sample collection were specified in the current NPDES permit and are summarized in Table 4.1 below. Eight of the nine stations were located within 150 m of the diffuser structure. The remaining station (1) was located 1,000 m north of the outfall, well beyond the discharge's sphere of influence. The reference station (1), midfield stations (2 and 7), and ZID[3] stations (4 and 5) have all been regularly sampled throughout the two decades of benthic monitoring. All lie along a

Table 4.1 Target Locations of Benthic-Sediment Sampling Stations

Station	Description	Latitude	Longitude	Distance (m) Closest[a]	Distance (m) Center[b]
1	Upcoast Reference	35° 23.730' N	120° 52.677' W	1,000	1,016
2	Upcoast Midfield	35° 23.280' N	120° 52.504' W	138	150
3	Upcoast Nearfield	35° 23.231' N	120° 52.504' W	49	60
4	Upcoast ZID	35° 23.210' N	120° 52.504' W	15	20
5	Downcoast ZID	35° 23.188' N	120° 52.504' W	15	20
6	Downcoast Nearfield	35° 23.167' N	120° 52.504' W	49	60
7	Downcoast Midfield	35° 23.118' N	120° 52.504' W	138	150
8	Offshore Nearfield	35° 23.199' N	120° 52.544' W	47	60
9	Onshore Nearfield	35° 23.199' N	120° 52.464' W	47	60

[a] Distance from the closest open diffuser port
[b] Distance to the center of the open diffuser section

north-south isobath (15.2 m) that intersects the center of the diffuser structure. In the October 1999 survey, four new stations (3, 6, 8, and 9) were added at nearfield distances (60 m) from the center of the diffuser structure. Two of the new stations (3 and 6) were located along the 15-m isobath, whereas the other two (8 and 9) were equidistant from the center of the diffuser structure in a cross-shore direction.

As described in Section 3.1.3 on Page 3-6, the finite length of the diffuser structure creates ambiguity in the distance measured to the outfall. For discussion purposes, station separations are usually referenced to the center of the diffuser. However, in terms of contaminant dispersion, the closest-approach distance is the controlling parameter.

Sampling at four new stations located close to the diffuser structure was intended to improve impact assessment. Assessments that rely solely on a comparison of sediment properties in a sample collected from the northernmost reference station (1) and properties averaged over stations close to the outfall can be misleading. Because of its 1-km separation from the outfall, the reference station has been found to have a measurably different infaunal community composition (MRS 1998bi), and as described below, an inherently different sediment physicochemistry. These differences are a consequence of its spatial separation alone. To ameliorate these zoogeographic differences, current impact assessments incorporate an evaluation of gradients close to the outfall, where any impairment would be expected to increase with proximity to the source.

Because the four new nearfield stations are configured in a symmetric pattern around the outfall, they stand a better chance of detecting impacts early, especially if preferential contaminant transport were to occur in a cross-shore direction. Also, by locating these new stations only 60 m from the outfall, they provide an early warning of any potential expansion of the discharge footprint. The existing '*gradient*' stations (2 and 7) were separated by three diffuser lengths from the discharge (150 m), and consequently, were much less likely to resolve benthic impacts related to effluent discharge. Assuming a uniform dispersion, a six-fold increase in contaminant discharge would have to occur before impacts would be detected at these two distant stations, as compared to the four new nearfield stations.

4.1.3 Benthic Sample Collection and Processing

Field sampling and laboratory procedures adhered to the requirements of MBCSD Monitoring and Reporting Program Number 98-15 (RWQCB-USEPA 1998ab). The 38-ft F/V *Bonnie Marietta*, owned and operated by Mr. Mark M. Tognazzini of Morro Bay, provided vessel support during the offshore survey. Dr. Douglas Coats of Marine Research Specialists (MRS) was chief scientist and was responsible for the collection of sediment chemistry samples. Ms. Bonnie Luke, also of MRS, was second science officer. Mr. Mark Tognazzini supervised vessel operations, and Mr. Marc Tognazzini assisted as deck hand.

Equipment

Sediment chemistry and benthic infaunal samples were collected using a chain-rigged Young grab whose design was modified from that of a Van Veen sediment sampler. The Young grab is equipped with a frame that enhances the penetration, stability, and the proficiency of the grab in collecting a level, undisturbed sample. Also, the stainless-steel grab is coated with Dykor®. Dykor® has properties similar to Tef-

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[3] The ZID is the "zone of initial dilution" as defined in the list of acronyms at the beginning of this report.

lon® and improves the chemical inertness of the grab sampler. This limits contamination of the sediment chemistry samples. The pair of jaws on the grab acquires sediments from a 0.1-m^2 area of the seafloor. During the October 2005 survey, a single grab sample was collected for sediment chemistry and bulk properties, and five replicate grab samples were collected for benthic infaunal analyses at each of the nine monitoring stations (Table D.1). At Station 1, an additional grab sample was also collected for the extraction of pore water. Pore water was extracted from the single chemistry grab samples collected at the remaining stations.

Prior to the field survey in 1999, the grab was completely refurbished. Rocks had damaged the jaws during an unrelated survey in early 1999 (Morro Group 1999). A stainless steel lip was also welded to the mouth of the jaw to improve the seal, which aids in sample retention during ascent. Finally, the entire grab was sandblasted and recoated with Dykor® in September 1999.

Offshore navigation aboard the survey vessel was supplied by a Furuno™ GPS 30 coupled to an FBX2 differential beacon receiver. Global positioning satellite (GPS) navigational fixes were recorded digitally at 1-second intervals. Electronic recordings of waypoints were used to mark the trip-time of the grab sampler and other pertinent information. Navigational errors inherent in standard GPS readings were greatly reduced with the use of a Differential GPS (DGPS) system that was first implemented by the U.S. Coast Guard. DGPS incorporates a second signal from a nearby land-based beacon. Because the beacon is fixed at a known location, the position error in the reading from the GPS satellites can be precisely calculated. This correction is continuously transmitted to the DGPS receiver and results in extremely accurate offshore navigation, typically with position errors of less than 2 m.

Physicochemical Samples

Prior to and during the survey, the sampling equipment was thoroughly cleaned to eliminate the introduction of contaminants into the samples and to prevent cross-contamination between stations. Before the survey, the grab sampler and sediment scoops were washed with Alconox®, deionized water, and 10% Hydrochloric Acid (HCl). During the survey, the grab and sampling utensils were washed with Alconox®, rinsed with seawater, and subsequently decontaminated with deionized water, methanol, and 1% HCl prior to sampling at each station.

Surficial sediments were collected from the upper 2 cm of a single grab sample recovered from each of nine benthic stations. Sufficient sample volumes were collected to provide material for QA/QC analyses. Samples were stored in the appropriate glass or plastic containers and refrigerated at approximately 4ºC prior to analysis. Chain of custody forms accompanied all sample shipments from the field and between laboratories. Appendix D.2 of this report contains the chains of custody and laboratory QA/QC analysis, while the other portions of Appendix D include a description of the samples and the chemical results as reported by the laboratory.

The sediments from each of the nine benthic-monitoring stations were separately analyzed for the analytes listed in Table 4.2. A summary of the analytical methods used and their authority is provided in Table D.2 of Appendix D. Analyses for other pesticides, hexachlorocyclohexane (HCH) isomers, cyanide, methylene chloride, and other volatile organics, are no longer performed because they have not been found in past sediment samples, are not elevated in effluent, and are not likely to accumulate within seafloor sediments.

Table 4.2. List of Analytes in Sediment Samples

• Sediment particle size	• Biochemical oxygen demand	• Total Kjeldahl nitrogen (TKN)
• Pore water dissolved sulfide	• Oil and grease	• Total volatile solids (TVS)
• Moisture content	• 11 trace metals	• Non-chlorinated phenolic compounds
• Chlorinated phenolic compounds	• Aldrin	• Dieldrin
• Chlordane	• DDT, DDE, DDD	• Endrin
• PAHs[4]	• PCBs[5]	• Toxaphene

Infaunal Samples

In addition to samples collected for sediment chemistry, five replicate grab samples were collected at each of nine benthic-monitoring stations for infaunal analyses. Upon retrieval of the grab, the sample was inspected for acceptability. Acceptance criteria were based on penetration depth, surface condition, and overall sample integrity. These observations are reported in Table D.1 in Appendix D of this report. An unusually high number of samples did not meet the minimum 7-cm penetration-depth criterion during the initial part of the 2005 survey. This required repeated drops with the grab sampler before a satisfactory sediment sample was collected for infaunal analysis. The unexpected difficulties with grab penetration arose because a higher-than-normal clay fraction was present within surficial sediments during the 2005 survey, as discussed in Section 4.2.1 on Page 4-15. Repeated sampling caused by low sample volumes was largely restricted to the first half of the survey at Stations 1 through 5. At those stations, sediment samples that met the acceptance criteria tended to have penetration depths close to minimum allowable depth (Table D.1). As the survey progressed, weights were added to the grab to enhance penetration, which mitigated the need for repeated sampling at the remaining stations.

Sediment samples that met the acceptability criteria were lightly washed, and elutriated onto a 1.0-mm mesh sieve. The extracted material was then washed into a labeled 16-oz. jar and preserved with 10% buffered formalin. After 48 hours, the formalin was rinsed from the samples on a 0.5-mm mesh sieve and the samples were transferred to 70% alcohol for processing, preservation, and storage. Samples were stained with a Rose Bengal solution to aid in sorting of the organisms into the following major taxonomic groups: annelida and nemertea, mollusca, crustacean, echinodermata, and miscellaneous phyla.

All individual organisms were enumerated and identified to species level where possible. Voucher collections have aided in the consistent identification of organisms collected throughout the 20-year benthic-monitoring program. The voucher collections contain specimens representative of each species identified, and are augmented as new taxa are collected. The qualifications and experience of the taxonomic specialists responsible for species identification are presented in Appendix E.1.

Sampling Location Offsets

As described in Chapter 3, there are a number of unavoidable factors that prevent collection of offshore samples at the exact target locations listed in Table 4.1 on Page 4-3. Figure 4.2 shows the actual position of the five infaunal (green) and additional chemistry (blue) grabs, relative to each target location (red) that was specified for Stations 2 through 9.

[4] Polynuclear Aromatic Hydrocarbons
[5] Polychlorinated Biphenyl

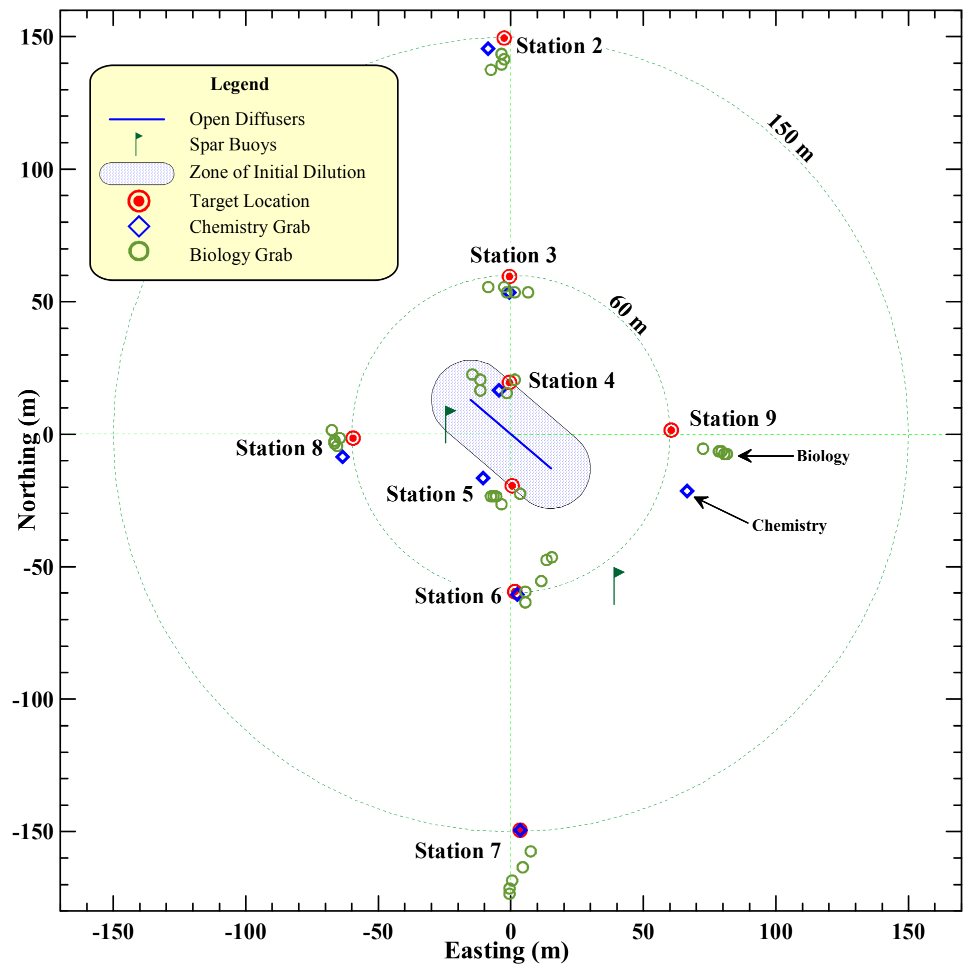

Figure 4.2 Actual and Target Locations of Benthic Monitoring Stations

Differences between the sampling and target positions arise because the vessel swings about the anchor location along a scope of approximately 20 m. The vessel set relative to anchor position is difficult to predict based on the constantly changing combinations of winds, waves, and currents. Changes in position during sampling at an individual station also arise because of these changing conditions, and because of anchor drag. These within-station position changes are evident in Figure 4.2 as dispersion among the in-faunal grabs shown in green. However, for a given station, position differences between the group of in-faunal (green) grabs and the chemistry (blue) grabs also arise because they were collected on two separate days. Consequently, some of the offsets shown in the Figure are due to differences in the anchoring locations on each day.

Despite the many unavoidable factors affecting vessel offset, differences between target position and average grab location were less than the 30-m width of the ZID. Table 4.3 lists the average position of the six grabs collected at each station. The largest offset from the target coordinates was 21 m at Station 1, and offsets of a similar magnitude occurred at Stations 7 and 9. For most of the stations however, the average offset was less than half of the offset observed at Station 1. Without the benefit of differential GPS navigation, most of these offsets would never have been resolved. Although the exact target locations

Table 4.3 Actual Locations of Benthic-Sediment Sampling Stations

				Distance (m)		
Station	**Description**	**Latitude**	**Longitude**	**Closest**[a]	**Center**[b]	**Offset**[c]
1	Upcoast Reference	35° 23.719' N	120° 52.675' W	979	996	21
2	Upcoast Midfield	35° 23.276' N	120° 52.505' W	129	142	8
3	Upcoast Nearfield	35° 23.229' N	120° 52.504' W	44	54	5
4	Upcoast ZID	35° 23.209' N	120° 52.508' W	10	20	7
5	Downcoast ZID	35° 23.187' N	120° 52.508' W	20	23	6
6	Downcoast Nearfield	35° 23.169' N	120° 52.499' W	43	56	9
7	Downcoast Midfield	35° 23.111' N	120° 52.505' W	152	164	15
8	Offshore Nearfield	35° 23.198' N	120° 52.548' W	53	66	6
9	Onshore Nearfield	35° 23.193' N	120° 52.454' W	61	77	19

[a] Distance from the closest open diffuser port [b] Distance to the center of the open diffuser section
[c] Average offset from Target Locations listed in Table 4.1

were not sampled, knowledge of the actual location where samples were collected allowed the spatial offsets to be factored into the gradient analysis.

4.1.4 Benthic Analysis

A wide variety of physical, chemical, and biological parameters were determined from the 54 sediment samples collected during the October 2005 survey. This report examines these parameters for any potential discharge-related spatial patterns. They are also combined with historical measurements to reveal any temporal trends that could be associated with a buildup of contaminants. Finally, the interrelation between biological and physicochemical parameters is evaluated for possible toxicological effects.

Organic-Loading Properties

Sediment samples collected at the nine benthic-monitoring stations during the October survey were analyzed for moisture content, oil and grease, eleven trace metals, and 100 synthetic organic contaminants. The organic-loading parameters, total volatile solids (TVS), total Kjeldahl nitrogen (TKN), and biochemical oxygen demand (BOD), are particularly relevant to sewage-outfall monitoring. If the volume of organic solids discharged by the outfall is not adequately dispersed, then a buildup of TVS, TKN, and BOD can occur beyond the ZID.

The total volatile solids (TVS) concentration quantifies the amount of organic matter in the solid fraction of bottom sediments. It is reported as percentage of dry-sample weight. Total Kjeldahl nitrogen (TKN) quantifies the amount of organic nitrogen present as ammonia, as well as that bound into organic compounds in the aqueous phase of the benthic samples. However, natural TKN levels vary widely among seafloor sediment samples, so anthropogenic (human-induced) impacts are difficult to discern from TKN measurements alone. Other properties related to organic loading from effluent discharge include sulfides, and oil and grease (O&G).

Biochemical oxygen demand (BOD) measures the depletion of dissolved oxygen caused by aerobic microorganisms during decomposition of organic material. Excessive BOD can be deleterious to marine organisms if decomposition significantly depletes dissolved oxygen levels. This can occur when aerobes metabolize excess organic matter deposited by effluent discharge. However, BOD is difficult to accurately measure in sediment samples, and interference from errant pieces of organic material has resulted in erroneously high levels in previous surveys, particularly prior to 1992 (Figure 4.9b on Page 4-29). Occasionally, though, interference still occurs. For example, BOD analysis of the sediment sample collected at Station 9 during the October 2000 survey exhibited marked interference, and it resulted in an erroneously high BOD (MRS 2001). Because of persistent interference in BOD determinations, overall confidence in BOD results for benthic sediment samples remains low.

Sulfide

Sulfide dissolved within pore water can be indicative of anoxic conditions and, in the form of hydrogen sulfide (H_2S), can be harmful to marine organisms. Sulfides are ionic compounds containing one or more sulfur ions bonded to carbon, metal, or other non-oxygen atoms. In marine sediments, sulfides exist both as insoluble precipitates and as dissolved sulfide compounds. In the presence of oxygen, sulfide rapidly oxidizes to sulfate, or in some instances, to elemental sulfur. Sulfides, therefore, are usually associated with hypoxic or anoxic conditions such as may occur in highly organic and undisturbed fine-grained sediments. Hydrogen sulfide (H_2S), the toxicologically important form of sulfide, is produced when bacteria reduce sulfates and putrefy proteins.

The analytical approaches used to measure sulfide levels within benthic sediments have evolved over the years of offshore monitoring. Consequently, long-term trends in sulfide concentration cannot be reliably determined from the dataset. Moreover, because of limitations in analytical approaches used prior to 2003, compliance with the sulfide standards specified in the COP[6] could not be unequivocally evaluated. In 2003, 2004, and 2005, an involved technique was implemented to extract pore water in the field. This technique produced aqueous samples that could be accurately analyzed for dissolved sulfide using EPA method 376.2. This resulted in high-resolution dissolved sulfide measurements that allowed direct evaluation of the COP limitation on "…*dissolved sulfide concentration of waters in and near sediments….*" Unfortunately, the flocculation and settling component of the technique requires comparatively calm offshore conditions, which may not always be available during future benthic surveys.

There were a variety of limitations associated with the analytical approaches used in prior years to measure sulfide. The Green and Schnitker (1974) method was used to determine sulfide concentrations in sediment samples collected prior to 2001. Not only was it difficult to properly implement this method in the field, but it led to large random variations in the measured sulfide levels. In addition to the complex sample-preparation in the field, the laboratory analysis required the use of specialized ion-specific electrodes. The practical quantification limit (PQL) was relatively high (2 mg/Kg), so sulfide was not detected in most samples. Occasionally, a comparatively high sulfide concentration would be reported in an isolated random sample, probably as a result of the presence of errant organic matter, such as a piece of detached kelp or sea grass, that happened to be included in the sediment sample.

In an attempt to stabilize the sulfide measurements, sediment samples collected in the 2001 and 2002 surveys were analyzed for sulfide concentrations using EPA Method 9030. Although the results were more reliable, they reported total sulfide concentrations, which as it turns out, were far different from the dissolved sulfides described in the COP. Because a large portion of the total sulfide concentration arises from acid-soluble metallic sulfides associated with particulate matter, reported concentrations are much

[6] The COP is the 1997 edition of the California Ocean Plan that applies to the current discharge permit for the MBCSD.

higher than those of dissolved sulfides. More importantly, they fail to track trends in dissolved sulfide concentrations, making evaluation of the dissolved-sulfide standard in the COP impossible. Moreover, a slight difference in the implementation of EPA Method 9030 caused a marked increase in 2002 sulfide concentrations compared to those of 2001. The increase was purely an artifact of the difference in laboratory digestion techniques, even though both analyses conformed to the method protocols.

In part, accurate measurement of dissolved sulfide in previous benthic surveys was hampered because sediment samples, rather than water samples, were being collected. Sediment samples do not lend themselves to chemical analysis methods designed for purely aqueous samples. The inappropriateness of the prior sulfide measurements was demonstrated by the analysis of the pore water samples collected during the 2003, 2004, and 2005 surveys. Dissolved sulfide concentrations were determined using EPA method 376.2, which has very low MDL of 0.05 mg/L. Dissolved sulfide was not detected in any of the 27 samples collected using the improved sampling technique.

The improved sampling technique was originally developed and applied for high-resolution trace-metal analyses that were to be conducted in an assessment of the marine bioavailability of chromite-mine contamination within Estero Bay. The potential influence of abandoned chromite mines on Estero Bay sediments is described later in this chapter. In preparation for the dissolved-metal investigation, interstitial pore-water samples were collected in 2003 by hand vacuuming using an aquarium air stone. This method has been found to be effective for the *in situ* collection of pore-water samples that were subsequently used to accurately define small gradients in dissolved constituents associated with shallow-water sediments (Sarda and Burton 1995). The technique was adapted for offshore sampling by working the air stone into a sediment sample while it was still contained within the grab sampler onboard the survey vessel. This technique was again used during the 2004 and 2005 surveys to extract a pore-water sample for dissolved-sulfide analysis. The pore water was extracted using a 60-mL syringe that was attached to the air stone via plastic tubing.

To avoid post-sampling increases in dissolved sulfide levels due to oxidation, the samples required preservation shortly after collection. However, suspended particulates in the sediment samples also had to be removed as part of EPA Method 376.2. Consequently, the pore-water samples were first treated with sodium hydroxide (HNaO) immediately after collection onboard the vessel. They were left to stand for approximately ten minutes with zero headspace in 250-mL containers while the suspended flocculates settled out. The supernate was then transferred to another container that held zinc acetate $[Zn(C_2H_3O_2)_2]$ as a preservative. In rough seas, this sampling procedure would not be possible because vessel motion induced by large waves would unduly agitate the pore-water samples, thereby hampering the timely settling of suspended particulates.

Synthetic Organic Compounds

The presence of synthetic organic compounds within sediment samples is often indicative of anthropogenic input and, in high enough concentrations, these compounds can be deleterious to the marine ecosystem. Phenolic compounds, which are used in plastics, adhesives, and coatings, are among the most common synthetic organic compounds measured in effluent. Benthic sediment samples were analyzed for phenolic compounds as well as for the pesticides aldrin, dieldrin, chlordane, DDT, endrin, and toxaphene. During the October 2005 survey, sediments were also analyzed for the toxic compounds PCB and PAH. As described in Chapter 2, few of these synthetic organic compounds have ever been detected in MBCSD effluent and, when they are, their concentrations are extremely low. Nevertheless, benthic sediments continue to be monitored for organic contaminants because a long-term discharge could conceivably result in a slow buildup over time, whereby sediment concentrations could eventually reach detectable levels. None of these synthetic organic compounds were detected in sediment samples collected at any of the stations during 2005, nor have any been detected in the sediment samples collected over the previous 19 years.

Trace Metals

In contrast to synthetic compounds, the presence of trace metals within seafloor sediments is not necessarily indicative of anthropogenic input. Most trace metals are found in detectable concentrations within naturally occurring mineral deposits, and some are even needed by marine organisms to survive. However, elevated levels of certain trace metals can be indicative of anthropogenic input, and excessive levels can cause deleterious effects in marine organisms.

Nine trace metals have been measured in seafloor sediments since the inception of the monitoring program. They are arsenic, cadmium, chromium, copper, lead, mercury, nickel, silver, and zinc. The toxicity of these metals to marine organisms has been examined in numerous laboratory bioassays and field experiments (Long and Morgan 1991, Long et al. 1995). These studies have been summarized in the form of biological effects levels, most notably an effects-range low (ERL), below which toxic effects are not expected, and an effects-range median (ERM), where adverse biological effects can be expected.

The benthic monitoring program described in the current NPDES discharge permit added aluminum and iron to the metals to be measured within sediments. Unlike the other nine metals, aluminum and iron are not particularly toxic to marine organisms, and their naturally occurring concentrations are relatively insensitive to anthropogenic input. The latter attribute makes them ideal tracers of natural variations in all the trace metals. Consequently, they have been used to normalize observed concentrations in the other nine metals so that background variation is reduced and anthropogenic trends can be revealed. In particular, iron has been used as a reference element for determining anthropogenic enrichment of trace metals within the Southern California Bight (Schiff and Weisberg 1997) and in many other offshore regions (Daskalakis and O'Conner 1995).

Physical Properties

Moisture content and grain-size distribution are two important physical properties of marine sediments. Moisture content is a measure of the volume of pore water present in the sediment samples. While not an indicator of discharge effects, it is an important parameter used in converting chemical concentrations measured in a wet sediment sample to a dry-weight basis. Dry-weight concentrations allow a more direct comparison of contaminant levels associated with bulk sediments by eliminating variations in sample mass that arise because of arbitrary differences in water content. Moisture content is also used, along with salinity measurements, to eliminate the bias introduced by salt content in grain-size determinations.

An increase in fine particulates near a municipal ocean outfall can be indicative of excessive deposition of particulates discharged by a treatment plant. In addition, the amount of fine sediment directly affects the composition of the infaunal community that resides within those sediments, although the precise mechanism for the relationship is rarely clear (Snelgrove and Butman 1994). Finally, natural variation in trace-metal concentrations has been correlated with the fine-sediment (mud) fraction and, like aluminum and iron, has been used to normalize metal concentrations (Dossis and Warren 1980, Ackerman et al. 1983, Horowitz and Elrick 1987).

In recognition of the potential influence of fine particulates, the previous NPDES discharge permit issued to MBCSD (RWQCB-USEPA 1993b) required that trace metal concentrations be normalized by the fine particulate fraction. However, this normalization was problematic because, in the case of the Estero Bay samples, the fraction of fine particles was very small. Normalization was found to amplify tiny random fluctuations in grain size, which produced erroneous contamination patterns. The current NPDES permit no longer requires normalization by the fine sediment fraction.

Nevertheless, without accurate measurement of the fine sediment fraction, variability in marine community composition or contaminant concentration could be erroneously ascribed to effluent discharge, when instead it may be the result of natural variation in the distribution of fine particulates. Because of the importance of fine sediments, the particle-size analysis was modified in 1998 from the standard pipette method described by Plumb (1981) and Folk (1980). The refined procedure produces significantly more accurate estimates of the fine grain-size fractions within marine benthic samples. An improvement to the method for determining fine grain-size fractions was necessary because of obvious inaccuracies in grain-size analyses conducted on sediment samples collected during past surveys. Specifically, an unusual bimodal grain-size distribution, where the smallest clay fractions exceeded silt fractions, had been consistently observed in sediment samples collected in the MBCSD monitoring program prior to 1998.

In 1994, the anomalous character of this size distribution led to a number of investigations and confirmatory grain-size analyses by outside laboratories (MRS 1995). The enigma was resolved in October 1997 after an exhaustive series of experiments using method blanks, spikes, and other diagnostic procedures. Findings indicated that the standard pipette analysis method overestimated the fraction of the finest-grained sediments in marine samples. As an aside, no differences were apparent in the inter-laboratory comparison study because the laboratories all apply the standard pipette procedure as part of their protocols. Consequently, they all overestimated the finest fraction as well. In contrast, the refined method currently in use for the MBCSD sediment samples accounts for the presence of dissolved solids (salt), as described by MRS (1998i) and Coats et al. (1999). As a result, the finest size fractions determined after October 1997 are far more accurate than those determined in prior surveys. Thus, some, if not all, of the long-term trends in grain-size database can be ascribed to improvements in methodology rather than actual changes in the benthic environment.

Biological Parameters

In the earliest monitoring surveys, analysis of biological samples produced two types of data on infaunal organisms: the number of organisms for each taxon, and the biomass of major taxonomic groups within each sample. However, the current NPDES permit no longer requires the determination of biomass because it is of little interpretive value, and its measurement is fraught with difficulty (MRS 1999b). Biomass data are difficult to interpret because the presence of large, undersampled organisms tends to severely skew results. Also, in practice, the heavy calcareous shells and exoskeletons of mollusks and crustaceans are troublesome to remove, and alcohol evaporation during weighing affects the determination of wet-weight biomass. Because of these problems, the USEPA does not recommend biomass for inclusion in 301(h) monitoring programs (USEPA 1987).

Pollution affects marine ecosystems by changing the number and type of benthic organisms found in the sediments. However, subtle changes in community composition are not always readily apparent in the large volume of raw data generated by a long-term monitoring program. Over the two decades of MBCSD monitoring, nearly 142,000 individual organisms representing 344 individual taxa have been collected. To assess whether the infaunal community has been impacted, these data must be summarized into concise parameters that are indicative of pollutant stresses. Biodiversity is a common indicator of the well-being of ecological systems, and forms the cornerstone of most impact assessment studies. Unfortunately, biodiversity is difficult to quantitatively define. Although a large number of diversity indices have been proposed over the years, most have been found to be of limited usefulness.

Diversity has two major components: species richness and evenness. Species richness measures the variety (number) of species, while evenness measures the distribution (abundance) of individuals among the species (Magurran 1988). Healthy ecosystems are thought to be rich in species that have an even distribution of individuals. However, no single measure can accurately represent changes in both evenness and richness along a pollution gradient. Nevertheless, many measures attempt to do so. The Shannon Diversity Index (H') is the most widely used diversity index because it exhibits some degree of sensitivity to both evenness and richness. In addition, most diversity indices only measure α-diversity, or the diversity within a local area at a particular time, *viz*., among replicate samples from a single station. In contrast, pollutant stresses tend to impact β-diversity, wherein community composition changes between samples separated spatially and temporally (Smith et al. 1979). Consequently, multivariate analysis techniques are better suited to the evaluation of β-diversity.

Most diversity indices are poorly suited for describing potential impacts to infaunal communities because they lack biological meaning, show little correlation with environmental quality, are difficult to interpret ecologically, and often result in ambiguous or biased estimates of diversity (Goodman 1975, Washington 1984, Green 1979). As a result, many indices are not recommended for routine inclusion in 301(h) monitoring programs (USEPA 1987). Nevertheless, the current NPDES permit requires that a number of infaunal community indices be reported and examined for temporal trends (RWQCB-USEPA 1998b). However, because of limitations associated with the indices, spatial differences with respect to outfall proximity, which are evaluated using standard statistical hypothesis tests, must be interpreted carefully. The indices computed in this report are listed in Table 4.4. The associated algorithms and guidance on how to interpret them are presented in Appendix E.2.

Table 4.4. List of Infaunal Parameters

• Number of individuals	• Number of species
• Number of individuals per species	• Species richness (S) (Magurran 1988)
• Margalef species richness (d) (Margalef 1951)	• Shannon-Wiener index (H')
• Simpson dominance (C') (Simpson 1949, Wittaker 1965)	• Brillouin index (h)
• Pielou evenness index (J') (Pielou 1977)	• Swartz dominance (S_w)
• Infaunal Trophic Index (*ITI*)	

4.2 MARINE SEDIMENT QUALITY

The quality of marine sediments surrounding the MBCSD outfall provides a definitive measure of potential marine impacts caused by the discharge of effluent. Specifically, discharge-related impacts have been demonstrated in other outfall monitoring programs, where marked changes in the benthic environment have been observed surrounding oceanic discharges located offshore of both central and southern California. For example, prior to improvements in effluent quality, contaminant buildup was observed in sediments surrounding large outfalls offshore southern California (Phillips and Hershelman 1996, Stull et al. 1986b). Another example comes from the continental shelf south of Estero Bay, where sediments exhibited small but statistically significant chemical signatures that were indicative of drilling mud discharges distributed over wide regions (Hyland et al. 1994, Coats et al. 1991, Coats 1994). If the MBCSD effluent discharge were causing significant impacts to the marine environment, similar changes in sediment properties would be evident near the diffuser structure. The challenge is in distinguishing changes that are caused by naturally occurring processes from changes that result from human activities.

Physicochemical changes in sediments provide a good indication of marine impacts for four main reasons. First, because outfalls are located on the seafloor, sediments are proximal to the source of potential contamination. Second, many contaminants are hydrophobic and tend to preferentially adhere to the fine particulates as they settle out of the effluent plume. Third, contaminants that are incorporated into seafloor sediments tend to have a long residence time because of the slow dispersive processes that prevail within pore waters. Finally, infaunal organisms that live within seafloor sediments are continuously exposed to any benthic contaminants because they cannot easily escape the source of pollution. If seafloor sediments or pore waters become polluted, then infaunal organisms, and ultimately the local marine ecosystem, can be impacted. Sedentary infaunal organisms provide a food source for other, more mobile organisms, such as finfish and shellfish. These trophic relationships can lead to bioaccumulation of contaminants within the marine food chain.

Despite the sensitivity of the benthos to effluent discharge, there has never been any indication of deleterious impacts to seafloor sediments as a result of the MBCSD discharge. This remained true during 2005, as it has in the previous 19 years of monitoring. The lack of negative benthic impacts is largely due to the comparatively low mass-emission of contaminants from the outfall. There has never been a large industrial contributor to the MBSCD influent stream, and the total discharge volume is small compared to ocean dischargers to the south, where discharge volumes can be over 100-times greater (e.g., SCCWRP 1995).

The lack of benthic impacts is evident from three sets of analyses performed on the chemical properties of sediment samples collected around the MBCSD outfall. Each of these three sets of analyses is discussed in the following subsections. First, concentrations of potential contaminants measured in sediment samples were below levels that have been identified as toxic to marine organisms. Second, there was no long-term trend in sediment concentrations that would suggest a buildup of contaminants close to the outfall. Finally, the spatial pattern of sediment concentrations near the outfall during 2005 bore no consistent relation to outfall proximity.

Although the concentrations of certain trace metals were found to be significantly different at the distant Reference Station 1, there were no perceptible spatial gradients at the other eight stations that were located at various distances close to the outfall. Also, the significant differences in concentrations at Station 1 were not consistent among the metals. For example, nickel and arsenic concentrations were lower at the Reference Station, while the chromium concentration was higher. Also, there were significant differences in the concentrations of naturally occurring sediment constituents at Station 1, such as aluminum and fine-particulate fractions. All of these aspects indicate that the anomalous conditions at Station 1 were due to natural spatial variation over the 1-km station separation between the reference station and the effluent discharge.

4.2.1 Sediment Contaminant Concentrations below Toxic Levels

In keeping with the California Ocean Plan (SWRCB 1997), the narrative receiving-water limitations in the Waste Discharge Requirements (RWQCB-USEPA 1998a) issued to the MBCSD state that the discharge should not cause chemical substances in marine sediments '… *to be increased above levels which would degrade indigenous biota*….' This requirement was established because contaminants that reside in sediments for long periods have the potential to exert acute and chronic effects on resident marine organisms (NOAA 1991ab, Finney and Huh 1989, Bertine and Goldberg 1977).

Over the 20 years of continuous benthic monitoring, there has never been any indication of an effluent-related degradation of marine organisms within Estero Bay. In large part, this is due to the consistently low contaminant concentrations within the discharged effluent (see Chapter 2). As a result, contaminant concentrations measured within Estero Bay sediments, both near and distant to the outfall, have been below thresholds considered harmful to marine biota. In fact, the comparatively pristine marine environment of Estero Bay is reflected in benthic contaminant levels that are generally well below concentrations measured in sediments offshore southern California.

Thresholds of Biological Effect

Over the years, marine monitoring programs have placed considerable emphasis on benthic contamination. This experience has resulted in a set of thresholds of biological concern that can be used to screen measured contaminant concentrations at individual sites. Specifically, these biological benchmarks can be used to assess whether the contaminant concentrations measured in sediments have the potential for adverse biological effects. Although the thresholds are not regulatory criteria, they provide toxicological guidelines that help assess the potential environmental significance of specific contaminant concentrations measured in the field.

The toxicological benchmarks advanced by NOAA (Long and Morgan 1991) are used in this report to evaluate contaminant levels measured during 2005. These guidelines were based on a correlation between chemical concentrations and observed biological effects in numerous modeling, laboratory, and field studies (Long et al. 1995). The studies identified two toxicological endpoints, an Effects-Range Low (ERL) and an Effects-Range Median (ERM) concentration. The lower toxicological endpoint (ERL) is associated with the concentration of a particular compound that produced an adverse biological effect in only 10% of the data. Below the ERL, adverse biological effects are unlikely. Above the ERM, which is based on the median of the toxicological data, adverse effects are likely to occur. The most conservative benchmark (ERL) was used in this report to determine the potential for marine biological impacts from most of the contaminant concentrations that were measured within Estero Bay during 2005. Because the ERL for nickel is ill defined by the toxicity studies, its ERM benchmark is used for comparison to measured levels instead.

Both trace-metal and organic contaminants are of concern in the benthic environment. However, their natural occurrence differs markedly. Synthetic compounds, such as PAHs, PCBs, and pesticides, are the organic constituents that can have the most significant anthropogenic impact if found in sufficient concentration. However, in this and many other marine sediment studies, their concentrations are below detection limits. In contrast, trace metals occur naturally in the marine environment and are commonly detected in sediment samples. Because of this, trace metals can serve as a powerful surrogate for evaluating the regional transport and fate of a host of particulate contaminants.

Grain-Size Distribution

Within deep offshore basins, sediment trace metals accurately record the history of contaminant accumulation because they are ubiquitous and long-lived. However, these deep benthic sediments often contain significant fractions of fine-grained sediments, which support an increased number of trace-metal bonding sites per unit of volume. Thus, large differences in trace metal concentrations among sediment samples can be caused by natural differences in mud fractions that are unrelated to human activities. Because of this, many marine benthic studies normalize trace metal concentrations by the mud, iron, or aluminum fraction. This reduces much of the inherent natural variability and, as a result, can reveal anthropogenic contaminant patterns normally masked by the natural association between metals and fine particulates (Dossis and Warren 1980).

In contrast, nearshore environments are subject to energetic oceanographic processes that erode fine particles, leaving behind coarser, more transport-resistant sandy sediments. These sandy sediments usually have comparatively low metal concentrations. Also, metals tend to be incorporated into the mineralogy of the sand grains rather than adsorbed onto particulate surfaces, as is the case with silts and clays. Because of this, metals are less likely to dissolve and become assimilated by marine organisms in sandy sediments. Because of the lower bioavailability of metals in sands, NOAA does not recommend normalization of trace-metals for samples with less than 20% fine-grained material. Such an approach would neglect a large proportion of contaminants that are bound within the sand-sized particles (Long and Morgan 1991). Estero Bay sediments consist almost entirely of sand (>99%, Table 4.5), and normalization grossly overemphasizes small inherent differences in the fine-particulate distribution (MRS 1998bh). Consequently, the monitoring requirements specified in the current NPDES permit no longer specify normalization of contaminant concentrations by fine sediment fractions (RWQCB-USEPA 1998b). In keeping with this, mud-normalized concentrations are not reported here.

Nevertheless, it is useful to examine the grain-size distributions independent of trace-metal analysis. For example, an increase in fine fractions close to the discharge point can be indicative of a buildup in effluent particulates within the surficial sediments. Also, the overall character of the grain-size distribution is indicative of the dynamics within a particular benthic environment. Increased mud fractions are indicative of quiescent depositional environments, while well-sorted sands result from energetic wave environments that winnow fine particulates as they constantly rework the surficial sediments.

Table 4.5 Grain-Size Distribution (%) in Seafloor Sediment Samples Collected on 5 October 2005

	Clay (9 to 12ϕ)	Silt (5 to 8ϕ)	Fine to Medium Sand (1 to 4ϕ)	Coarse Sand (1ϕ)	Very Coarse Sand (0ϕ)	Gravel (-1ϕ to –3ϕ)
Station 1	0.41	0.50	98.88	0.06	0.04	0.11
Station 2	0.35	0.33	98.82	0.42	0.04	0.04
Station 3	0.34	0.32	98.82	0.52	0.01	0.00
Station 4	0.34	0.28	99.03	0.27	0.06	0.02
Station 4[a]	0.42	0.22	98.87	0.46	0.03	0.00
Station 5	0.38	0.27	98.86	0.46	0.04	0.01
Station 6	0.29	0.31	99.01	0.38	0.01	0.00
Station 7	0.39	0.38	98.64	0.58	0.02	0.00
Station 8	0.32	0.53	98.70	0.43	0.03	0.00
Station 9	0.32	0.28	98.91	0.46	0.03	0.00
Station 9[b]	0.34	0.51	99.08	0.07	0.01	0.00
Mean[c]	**0.35**	**0.35**	**98.88**	**0.34**	**0.03**	**0.00**

[a] Duplicate analysis of Station-4 sediment sample
[b] Duplicate analysis of Station-9 sediment sample
[c] Mean computed after arc-sine transformation

The statistical analyses of grain-size distributions are summarized in Appendix Table D.4. The Table shows that Estero Bay sediments consist of well-sorted fine sands with a distribution skewed slightly toward the finer fractions. Mean particle diameters range from 142 μm to 164 μm and lie in the middle of the fine-sand portion of the grain-size distribution. The high degree of sorting is reflected by low standard deviations (σ<0.381). The small, but consistently positive skew to the distribution reflects a slight excess in the finer fractions relative to larger fractions. Together, these properties point to a highly energetic flow regime that tends to winnow fines from the surficial sediments. The contribution of larger-diameter fractions was negligible in that the combined coarse sand and gravel fractions (1 to -3ϕ) represented less than

0.6% of the sample weight (Table 4.5). These largest fractions consisted mainly of fragments of shell hash that are randomly distributed on the surface of sediments in many shallow subtidal coastal areas.

The clay fraction in sediment samples that were collected during the 2005 survey was much larger than in previous surveys. The clay fraction was uniformly high at all stations indicating that the increase in fine sediments was due to regional processes rather than increased particulate deposition from the MBCSD discharge (Table 4.5). The mean clay fraction (0.35%) in 2005 was 2.5-times higher than the mean clay fraction in 2004, and more than double the largest fraction found in surveys conducted since 1999. Although it is not clear what is responsible for the recent buildup in this finest sediment fraction, it could be caused by a decrease in wave-induced reworking of surficial sediments or by an increase in the input of terrigenous particulates within runoff.

Undetected and Nontoxic Metals

In 2005, trace-metal concentrations within sediment samples were either not detected or were below levels that are potentially toxic to marine organisms (Table 4.6). Cadmium, mercury, and silver were not detected at quantifiable levels within the sediments, as has been the case in most prior monitoring surveys. The quantification limit for cadmium (PQL=0.16 mg/Kg-dry) was much smaller than the lowest toxic-effects level (ERL=1.2 mg/Kg). Similarly, silver's PQL (0.07 mg/Kg-dry) is 15-times lower than its lowest threshold of biological effects. Although the quantification limit for mercury (0.05 mg/Kg-dry) is only three-times lower than its ERL, the ERM, where biological impacts are expected, is much higher, near 0.71 mg/Kg. Thus, even if these three metallic elements were present in Estero Bay sediments at concentrations below their respective PQLs, they would not be likely to affect marine organisms.

Two of the remaining eight metals, namely aluminum and iron, are not toxic to marine organisms. The concentrations of those two metals quantify changes in the concentrations of the other trace metals that occur naturally within the sediment samples because of inherent differences in mineralogy. The other six metals have been consistently detected in Estero Bay sediments throughout the history of the benthic-monitoring program. Although they have the potential for anthropogenic enrichment, their mean concentrations have remained comparable to ambient levels for the region, and below (ERM) levels likely to be toxic to marine organisms. In 2005, as in previous years, five of these six metals had concentrations well below the ERL where biological effects begin to become apparent. Only the mean nickel concentration was above the ERL. Each of these six detected metals is described below.

Chromium

Historically, most of the chromium detected in the Estero Bay sediments was in a trivalent form that occurs naturally as chromite ore. This is evident from the lack of measurable hexavalent chromium levels in samples collected prior to 1999 (MRS 1996, 1997c, 1998bi, and 1999b). Industrial processes generally produce the hexavalent and metallic oxidation states of chromium. Although chromite ore is a widely distributed mineral deposit, it is more prevalent along the south-central coast of California (SAIC 1986). Naturally occurring chromite is introduced to the Central Coast's waters by fluvial transport of eroded ultrabasic minerals found in the Franciscan formation. The Franciscan formation outcrops along the headlands north of Point Estero and in the Santa Ynez Mountains.

Table 4.6 Metal Concentrations (mg/Kg-dry) in Seafloor Sediment Samples Collected on 5 October 2005

Station	Aluminum	Arsenic	Cadmium	Chromium	Copper	Iron	Lead	Mercury	Nickel	Silver	Zinc
1	6700	3.6	≈0.17 [a]	62	3.7	10,400	2.4	<0.05 [b]	37	<0.07	17
2	6300	4.2	≈0.19	53	4.8	10,700	2.9	<0.05	50	<0.07	19
3	5900	4.2	<0.16	54	4.3	9,800	2.2	<0.05	47	<0.07	18
4	5300	4.4	<0.16	45	3.7	8,900	2.0	<0.05	39	<0.07	15
5	5200	4.2	<0.16	43	3.7	8,700	2.1	<0.04	41	<0.07	14
6	6000	4.3	<0.16	43	3.6	10,200	2.3	≈0.07	43	<0.07	15
7	6100	4.4	≈0.15	47	4.1	10,200	2.2	<0.04	45	<0.07	16
8	6300	4.4	≈0.17	52	4.3	10,500	2.4	<0.05	48	<0.07	16
9	5400	4.4	<0.16	45	3.8	9,100	2.1	<0.05	42	<0.07	15
Mean[c]	**5900**	**4.2**	**<0.17**	**49**	**4.0**	**9,800**	**2.3**	**<0.07**	**43**	**<0.07**	**16**
ERL	—[d]	**8.2**	**1.2**	**81**	**34.0**	—	**46.7**	**0.15**	**20.9**	**1.0**	**150**
ERM	—	**70.0**	**9.6**	**370**	**270.0**	—	**218.0**	**0.71**	**51.6**	**3.7**	**410**
SCB[e]	—	**6.2**	**0.2**	**28**	**11.7**	—	**10.3**	—	**29**	**0.2**	**44**

[a] The metal was detected but the reported concentration was not well quantified. Namely, the reported concentration exceeded the method detection limit (MDL) but was below the PQL.

[b] The metal was not detected above the indicated MDL

[c] After logarithmic transformation [d] Guideline not specified

[e] Concentration that would be considered enriched relative to background concentrations in the Southern California Bight based on normalization by iron (Schiff and Weisberg 1997).

The influence of these local sources of chromite ore is evident in the high total chromium concentrations measured in Estero Bay relative to background concentrations observed offshore southern California (Schiff and Weisberg 1997). An average ambient chromium concentration near 17 mg/Kg was found in sediment samples collected within the Southern California Bight that contained approximately 1% iron, which is close to the average iron concentration in the Estero Bay samples (Table 4.6). At 49 mg/Kg, average chromium concentrations within Estero Bay were much higher than the Bight average, and were approximately 75% higher than the concentration (28 mg/Kg) considered to be enriched in the Bight. Only approximately 25% of the Bight had sediments with chromium concentrations higher than those measured in Estero Bay. Most of this area was within Santa Monica Bay, where a high mass-emission of contaminants from POTWs[7] occurred before 1988. In contrast, the slightly elevated ambient chromium concentrations found in the Santa Barbara Basin in the northern reaches of the Bight have been ascribed to the presence of naturally occurring chromite ore in the region (Chow and Earl 1978).

The fact that chromium also occurs naturally in the mineralogy of the Central Coast explains its overall enrichment within the marine sediments of northern Estero Bay. However, ordinary erosional processes cannot account for the observed increase in chromium concentrations that has occurred for at least the last two decades. Long-term increases in chromium, arsenic, copper and nickel are documented in Section 4.2.3. Although these metals are naturally enriched in onshore mineral deposits, nearshore depositional sediments should have reached equilibrium with onshore sediments that have been eroded within the ad-

[7] Publicly-Owned Treatment Works

jacent watershed throughout geologic time. Instead, the increases in marine sediment concentrations that have been observed throughout the monitoring program suggest that the erosion of these mineral deposits has been accelerated by human activities, such as mining. As described by MRS (2000b, 2001), sediment samples collected in conjunction with a supplementary environmental project indicate that erosion around abandoned chromite mines is the likely cause for the increases observed in the marine sediments of Estero Bay.

Regardless of its source, chromium concentrations in all sediment samples collected during 2005 remained below the lowest toxic level (ERL of 81 mg/Kg in Table 4.6) where biological effects might first be observed. The highest concentration (62 mg/Kg) was measured at Station 1. It was just over one-sixth of the concentration at which biological effects become probable (ERM = 370 mg/Kg). Although the chromium effects levels were reliably determined, they are not '*highly predictive of adverse biological effects*,' as stated by Schiff and Gossett (1998) in their review of contaminant concentrations within the Southern California Bight. On the contrary, Long et al. (1995) state that the incidence of effects in the toxicological studies used to establish the levels for chromium were '*greatly influenced and exaggerated by data from multiple tests conducted in only two field surveys*.' In any regard, because measured concentrations within Estero Bay sediments during 2005 were below the lowest effects level, it is unlikely that chromium caused any adverse effects on marine organisms.

Arsenic

Arsenic is a highly poisonous metallic element used in insecticides, herbicides, solid-state doping agents, and various alloys. Arsenic is also present in naturally occurring substances and is widely distributed throughout nature. As a result, mine tailings may contain high residual arsenic levels due to the presence of arsenopyrites in the ore, overburden, and soil. Depending on the location and composition of the mine overburden, arsenic may be released into surface and ground waters through erosion.

Regardless, mean arsenic concentrations in Estero Bay sediments were just over half of the lowest toxicity level (ERL of 8.2 mg/Kg) where adverse biological impacts would first begin to appear. They were an order of magnitude smaller than the ERM (70 mg/Kg) where biological impacts become probable. For comparison, approximately 75% of the area within the Southern California Bight has arsenic concentrations exceeding the mean levels measured in Estero Bay during 2005 (Schiff and Gossett 1998). After adjustment for the mean iron concentration, average arsenic levels would have to be 2 mg/Kg higher to be considered enriched relative to background concentrations in marine sediments to the south (SCB = 6.2 mg/Kg in Table 4.6).

Copper

As with arsenic and chromium, copper commonly occurs in Central-Coast minerals. The average copper concentration (4.0 mg/Kg) in marine sediments collected at the nine benthic-monitoring stations in 2005 was well below the level that that would be considered enriched relative to background concentrations in sediments to the south (11.7 mg/Kg). All of the sediment samples had measured copper concentrations that were also far below levels that would begin to affect marine organisms (ERL=34 mg/Kg). In contrast to chromium effect levels, the biological effect levels of copper are well defined by toxicological studies. The highest measured copper concentration was 50-times lower than the ERM level where adverse effects become probable. The amount of copper in Estero Bay sediments is also negligible compared to marine sediments offshore southern California, where approximately 90% of the Southern California Bight contains sediments with higher copper concentrations.

Lead and Zinc

Lead and zinc concentrations in Estero Bay sediments are low compared to ERL levels where adverse biological effects first become noticeable. They are also low compared to concentrations within the Southern California Bight, where approximately 95% of the area has higher concentrations. Most of the elevated lead and zinc concentrations to the south were measured within Santa Monica Bay and on the Palos Verdes Shelf, where large POTWs have discharged for many decades. However, most of the mass emissions of metals from these treatment plants occurred prior to the substantial improvements in effluent quality that were realized in 1988. Apparently, the historical discharges from these outfalls have accumulated within adjacent sediments and have persisted for decades (Stull et al. 1986b). In contrast, the historically low emission of contaminants from the MBCSD treatment plant has had no measurable effect on the high sediment quality within Estero Bay.

Nickel

Although unrelated to wastewater discharge, nickel was the only trace metal whose concentration in Estero Bay sediments consistently exceeded its ERL level (Table 4.6). In fact, in the two decades of benthic monitoring, only one sample has contained nickel in concentrations less than the ERL of 20.9 mg/Kg. In 2005, the nickel concentrations at Stations 2, 3, and 8 even approached the ERM level (51.6 mg/Kg) where adverse effects are normally considered probable. Despite measured nickel concentrations that exceed the lower marine-toxicological benchmark, there were no apparent deleterious impacts to infaunal organisms living within northern Estero Bay sediments. There are two possible reasons for this.

First, nickel exhibits a very weak relationship between the incidence of effects and concentrations in the database used to establish the toxic-effect ranges (Long et al. 1995). Because of this weak toxicological relationship, specification of bulk nickel concentrations that induce adverse reactions in marine biota is highly uncertain. Much of this uncertainty arises from wide variability in nickel bioavailability. When nickel is in a dissolved form, or adheres to surfaces of fine-grained sediments, it is much more likely to impact organisms that ingest or come into contact with the sediments. Conversely, nickel that is bound into the mineralogy of larger sand grains has less influence on marine organisms. The sandy sediments within the MBCSD benthic monitoring area fall into the latter category.

The second reason for the lack of impacts from elevated nickel concentrations is that nickel occurs naturally in the mineralogy of the southern Central California Coast (Steinhauer et al. 1994). Nickel and chromium often co-occur in mineral deposits, and it is likely that the increased nickel concentrations found offshore also derive from the Franciscan geologic formation that outcrops at Point Estero. Thus, much of the measured nickel is bound into the mineral matrix of the sand grains themselves, where marine fauna cannot readily metabolize it. Even if there is a slightly higher dissolved-phase nickel concentration within the pore waters of Estero Bay, benthic organisms may have adapted to the elevated nickel concentrations that have probably been present for epochs.

Nickel and chromium are the only metals whose measured bulk concentrations in Estero Bay sediments consistently exhibit '*anthropogenic enrichment*' relative to background concentrations in the Southern California Bight (Schiff and Weisberg 1997). However, this designation is not completely correct because much of the regional enrichment arises from natural erosional processes, rather than from accelerated erosion of contaminated mine tailings. In fact, the influence of regional mineralogy is also plainly evident in the Southern California Bight database where '*background*' nickel concentrations steadily increase from south to north. Schiff and Gossett (1998) did not account for this natural variability in ore deposits and erroneously included nickel trends in their assessment of benthic contamination within the Southern California Bight.

Organic Pollutants

Benthic sediments were also analyzed for synthetic organic compounds, including chlorinated phenolic compounds, non-chlorinated phenolic compounds, PCBs, PAHs, and pesticides. None of the 100 measured compounds were detected at any of the nine stations (See Appendix D.1). PQLs for all the organic compounds were well below ERL guidelines where marine biological impacts become likely for these compounds.

There are no well-established marine biological effects levels for organic-loading parameters such as TKN, BOD, TVS, and O&G (Table 4.7). Moisture content, mud fraction, TKN, BOD, and TVS measured in the sediment samples collected in 2005 were comparable to levels measured in the past five years. As described in Section 4.1.4 on Page 4-8, the dissolved-sulfide analyses conducted on pore-water samples in 2005 represent a significant improvement over previous sulfide analyses that were conducted on bulk sediment samples. As a result, the 2005 measurements provide the best evaluation of compliance with the sulfide limitation in the COP. No dissolved sulfides were detected in the survey, even though the detection limit was very low (MDL=0.05 mg/L).

Table 4.7 Non-metal Concentrations (mg/Kg-dry)[a] in Seafloor Sediment Samples Collected on 5 October 2005

Station	Moisture (%)	Dissolved Sulfides (mg/L)	BOD	Oil & Grease	TKN	Volatile Solids (%)
1	24.1	<0.05	37	<36	132	1.71
2	24.4	<0.05	41	<36	132	1.98
3	23.7	<0.05	31	<35	114	1.83
4	24.8	<0.05	35	<36	116	1.73
5	23.8	<0.05	41	<35	96	1.71
6	25.3	<0.05	47	<36	116	1.87
7	24.2	<0.05	37	<36	123	1.72
8	25.1	<0.05	40	<36	187	2.00
9	24.4	<0.05	33	<36	115	1.85
Mean[b]	**24.4**	**<0.05**	**38**	**<36**	**124**	**1.82**

[a] Unless otherwise indicated
[b] Arc-sine transformation for % and logarithmic transformation for mg/Kg

4.2.2 Contaminant Distribution Unrelated to Outfall Proximity

In 2005, as in previous years, contaminant distributions failed to exhibit a spatial pattern indicative of a benthic environment that was impaired by discharged effluent. Specifically, there was no consistent evidence that sediment contaminant concentrations near the diffuser structure were elevated relative to distant sites. A gradient of steadily decreasing contaminant concentrations with increasing distance from the outfall would be *prima facie* evidence of discharge-related impacts in the marine environment. This pattern of impairment has been observed around larger outfalls servicing more industrialized locales (Phillips and Hershelman 1996, Stull et al. 1986b, Stull 1995, Diener et al. 1995). The lack of contaminant loading around the MBCSD outfall was confirmed through application of rigorous statistical hypothesis tests and by qualitative comparisons of concentration levels ranked by distance from the diffuser structure.

During any given year, an isolated constituent may exhibit an anomalous value at one station that the spatial analysis interprets as a statistically significant gradient at the 95% confidence level. However, by the very definition of the 95% confidence level, one in twenty tests would be expected to reveal a "*significant*" gradient when, in fact, none exists. Over the years, many constituents have been tested for spatial gradients, and occasionally the analysis identifies one that may be statistically significant at some particular time. Upon further analysis, however, the gradient has always been found to be unrelated to the discharge. Approximately half of the time, these gradients display a pattern of decreasing contamination with increasing proximity to the outfall, which is opposite of that expected from effluent discharge. For example, in 2005, statistically significant gradients ($p<0.02$) in arsenic and chromium concentrations arose because of a difference in the physical environment at Station 1, which lies 1 km from the outfall (Figure 4.3). The significantly higher concentration of chromium that was observed at the reference site produced a spatial gradient that is opposite of that which would be generated by contamination from wastewater discharge. Similarly, it was only the lower concentration of arsenic at Station 1 that led to a significant spatial gradient. There was no spatial gradient in arsenic concentrations among the other stations ($p=0.94$).

The bar graphs in Figure 4.3 show trace-metal concentrations ranked by distance from the diffuser. Consistent spatial gradients are not visually apparent in the bar graphs. Instead, most concentrations vary randomly about the survey mean within the 95% confidence interval (shown to the left of each bar graph).

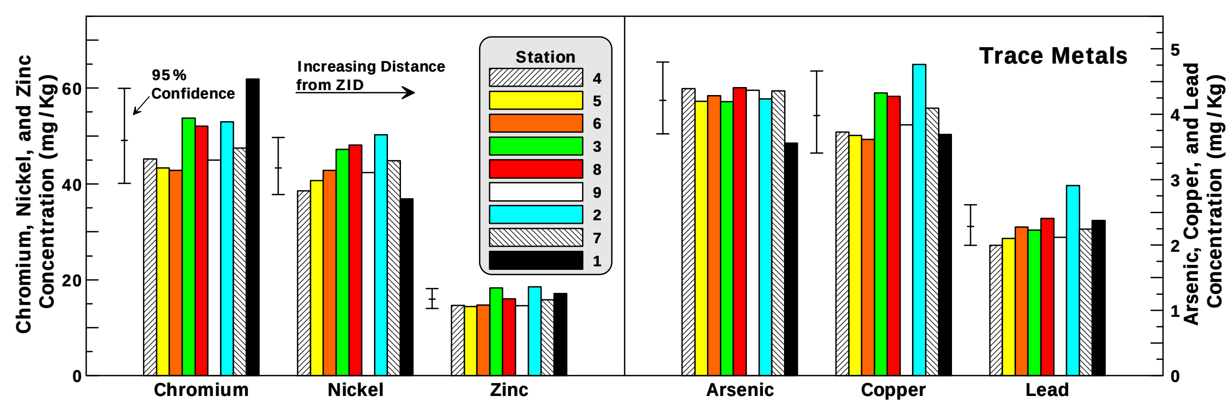

Figure 4.3 Trace-Metal Distribution in Sediment Samples Collected during 2005

This demonstrates that the differences in concentration measured in samples collected from individual stations are largely due to small-scale sampling variability rather than to actual differences in contaminant loads across the survey area. The confidence limits reflect the within-station replicate variability determined from three replicate chemistry samples that were collected and separately analyzed at each station in 1993 and 1994. Thus, they provide a measure of the inherent within-station sampling error for each of the parameters. If this within-station variability is larger than the differences in concentration between stations, then there is a low degree of confidence in any perceived spatial differences between stations. Figure 4.3 shows that this was the case for most trace-metal concentrations measured in 2005.

Only the chromium and arsenic concentrations at reference Station 1, and copper and lead concentrations at Station 2, extend well beyond the 95% confidence limits. However, these locally anomalous trace-metal concentrations, as well as the smaller differences among other stations, can be attributed to natural processes. This is apparent because the variability in aluminum and iron concentrations, shown in Figure 4.4, tends to track the spatial distribution of many of the anthropogenic metals shown in Figure 4.3. In particular, the highest aluminum concentration was observed at Station 1, while the highest iron concentration was measured at Station 2.

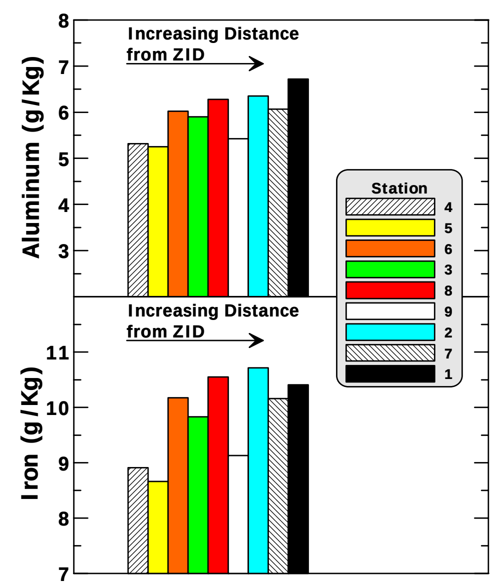

Figure 4.4 Aluminum and Iron Distributions in Sediment Samples Collected during 2005

Both aluminum and iron are good indicators of natural trace-metal variability because anthropogenic activities have little influence on them, and they are not enriched in the MBCSD discharge. Instead, increased concentrations in these two metals reflect a more quiescent depositional environment where fine sediments are preferentially deposited by natural processes. Normalization of raw trace-metal concentrations by aluminum, iron, or mud concentrations is common practice in impact-assessment studies because it can reveal anthropogenic patterns that are otherwise masked by variability in background concentrations.

Accordingly, the spatial variability in trace-metal distributions shown in Figure 4.3 is due to natural processes. For example, the differences in raw chromium, nickel, and zinc concentrations among stations (left frame of Figure 4.3) is markedly reduced after normalization by aluminum (top left frame of Figure 4.5). Similarly, the particularly high lead and copper concentrations at Station 2 (right frame of Figure 4.3) is tempered after normalization by iron (bottom right frame of Figure 4.5). Normalization by aluminum reduced the coefficient of variation (CV) in chromium, nickel, and zinc concentrations by a factor of more than ten. At the same time, normalization did not introduce any new trends indicative of an impact signature wherein contaminant concentrations consistently increase with proximity to the outfall.

Statistical hypothesis testing of differences between mean concentrations at nearfield and far-field stations provides another quantitative method for detecting potential discharge-related gradients (Table 4.8). Specifically, a statistical test was performed on the null hypothesis that the mean chemical concentration within 61 m of the diffuser structure was not significantly higher than the mean concentration among midfield and reference stations (Stations 1, 2, and 7). The test compared the magnitude of the difference in mean concentrations with the variability about those means. If the variability is high, then even large differences in mean values may not be significant.

Table 4.8 shows that the null hypothesis could not be rejected for any of the raw metal concentrations or organic loading parameters. In fact, computed mean concentrations were higher at distant stations for all constituents except arsenic and TVS. Sediment properties that have an increased concentration at distant sites are distinguished by a negative *t*-value and brackets around the *p*-value in Table 4.8. Obviously, a higher contaminant concentration at distant sites is a pattern opposite of that expected in a benthic environment impaired by effluent discharge. In the case of aluminum and lead, this spatial difference would

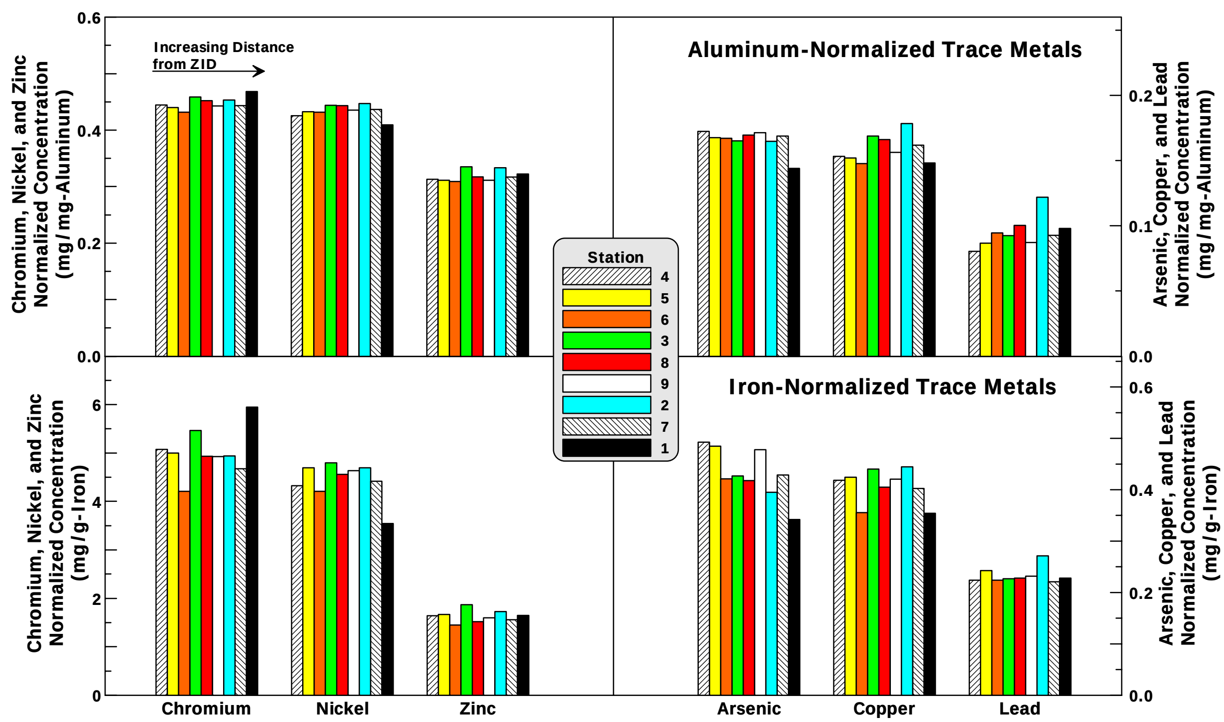

Figure 4.5 Aluminum-Normalized and Iron-Normalized Trace-Metal Distributions in Sediment Samples Collected during 2005

have been determined to be "*statistically significant*" at the 95% confidence level, had it not been in the wrong direction. As shown by *p*-values in the two columns on the right-hand side of Table 4.8, normalization by iron and aluminum did not perceptibly alter the results of the hypothesis tests for spatial gradients related to outfall proximity. Only normalization by iron increased the concentration difference in arsenic enough to reject the null hypothesis (p=0.02). However, the enhanced difference was due to the much lower iron concentrations found at the ZID Stations 4 and 5 (bottom frame of Figure 4.4) rather than any actual difference in the raw or aluminum-normalized concentrations at those stations (Figure 4.3 and the top right frame of Figure 4.5).

As with trace metals, differences in organic-loading parameters among stations were small compared to overall replicate variability (Figure 4.6). The 95% confidence intervals generally encompass the range of concentrations measured at each station. The anomalously high TKN concentration at Station 8 is the only noteworthy exception. The higher TKN concentration at this station is probably related to the higher mud fraction that was also found at this station. In any regard, there were no trends in organic loading parameters where concentrations regularly increased with outfall proximity (p>0.4). The mud fraction shown in the lower right-hand bar graph was the only parameter that exhibited a perceptible spatial gradient in concentrations in relation to outfall proximity (p=0.024). However, the apparent increase in mud fractions at distant stations is a trend opposite of that induced by effluent-discharge impacts. Moreover, this spatial gradient was largely driven by markedly higher mud fractions that occurred only at Stations 1 and 8, rather than a trend of steadily increasing fine fractions among the other stations.

Table 4.8 Tests for Significant Increases in Contaminant Concentrations near the Diffuser

Parameter	Mean (mg/Kg) Distant[a]	Near[b]	t[c]	p[d]	Iron-normalized p	Aluminum-normalized p
Aluminum (g/Kg)	6.37	5.69	-2.37	[0.02]	[0.12]	—
Arsenic	4.03	4.31	1.51	0.09	**0.02**	0.06
Chromium	53.8	46.8	-1.80	[0.06]	[0.25]	[0.11]
Copper	4.16	3.90	-0.96	[0.19]	0.35	[0.25]
Iron (g/Kg)	10.4	9.5	-1.89	[0.05]	—	0.08
Lead	2.49	2.18	-2.03	[0.04]	[0.19]	[0.05]
Nickel	43.7	43.1	-0.15	[0.44]	0.13	0.32
Zinc	17.1	15.4	-1.68	[0.07]	[0.41]	[0.13]
BOD	38.2	37.4	-0.23	[0.41]		
TKN	129	121	-0.45	[0.33]		
TVS (%)	1.80	1.83	0.35	0.37		
Mud (%)	0.78	0.66	-1.74	[0.06]		

[a] Mean concentration computed among Stations 1, 2, and 7, which lie at a distance of 129 m or more from a diffuser port (see Table 4.3 and Figure 4.2). An arc-sine or logarithmic transform was applied as appropriate.

[b] Mean concentration computed among Stations 3, 4, 5, 6, 8, and 9, which lie within 61 m of a diffuser port

[c] *t*-value resulting from a comparison of mean concentrations relative to measurement variability. A negative *t*-value indicates that the computed mean concentration was lower at stations near the diffuser, opposite of that expected in an environment impacted by effluent discharge.

[d] *p*-value representing the probability that a higher nearfield mean concentration is not significant. In this study, *p*-values less than 0.05 (95% confidence) indicate that nearfield means are significantly higher, and the null hypothesis can be rejected. Note, however, that a '*one-sided*' *t*-test was performed, in which only higher nearfield concentrations are meaningful. Consequently, *p*-values for negative *t*-values are not pertinent and are indicated with brackets.

[e] Hypothesis tests performed on trace-metal concentrations normalized by the iron concentration to reduce the potential influence of inherent variability in background metal concentrations.

As with aluminum and iron concentrations, the increased mud fractions at Stations 1 and 8 indicate that the benthic environment was more quiescent at these stations as compared to the other monitoring stations. The percentage of the finest sediment particulates at these stations was slightly higher than the average mud fraction. The increase was within the 95% confidence interval shown in Figure 4.6, and accordingly, the null hypothesis that average mud concentrations were similar at near and distant stations could not be rejected, but the probability was only marginal (p=[0.06] in Table 1.8). In contrast, linear regression on the mud spatial gradient was found to be statistically significant (p=0.024) as mentioned above.

An increased fine fraction at Station 8 is expected because of its increased depth and distance from shore. Station 8 is located 60 m offshore of the diffuser structure in a water depth that is approximately a meter deeper than the isobath along which most of the other stations are located (Figure 4.2 on Page 4-7). Because wave-induced resuspension and scour decrease with increasing water depth, Station 8 retains a greater proportion of fine sediments in its more-quiescent benthic environment. Similarly, Station 1 is located 1 km north of the diffuser structure, making it more than seven times farther from the diffuser structure than the other stations (Figure 4.1 on Page 4-2). It, too, appears to be located in a more quiescent benthic environment.

4.2.3 Long-Term Contaminant Buildup Unrelated to Effluent Discharge

The previous section examined the spatial distribution of constituents at a single point in time, namely, October 2005. However, as described above for Stations 1 and 8, ambient sediments at individual stations can be inherently different due to natural processes. These inherent differences complicate the interpretation of any perceived spatial gradients with respect to effluent discharge. On the other hand, examining changes in spatial distributions over time significantly improves the power to detect potential discharge-related impacts. Specifically, if there were any measurable marine impacts from wastewater discharge, they would be revealed by a long-term buildup of contaminants in sediments surrounding the outfall. This is because particulates, which carry those contaminants, tend to be preferentially deposited near the point of discharge (the outfall diffuser). Consequently, examining spatial differences in contaminant concentrations over time can reveal discharge-related trends that are completely hidden in a spatial analysis of a single survey.

As revealed by the temporal analyses described in this section, there has been a long-term buildup in some sediment contaminants. However, these concentration increases have occurred consistently at all stations. Consequently, large-scale processes unrelated to effluent discharge are driving the ambient contaminant distributions within the sediments of Estero Bay.

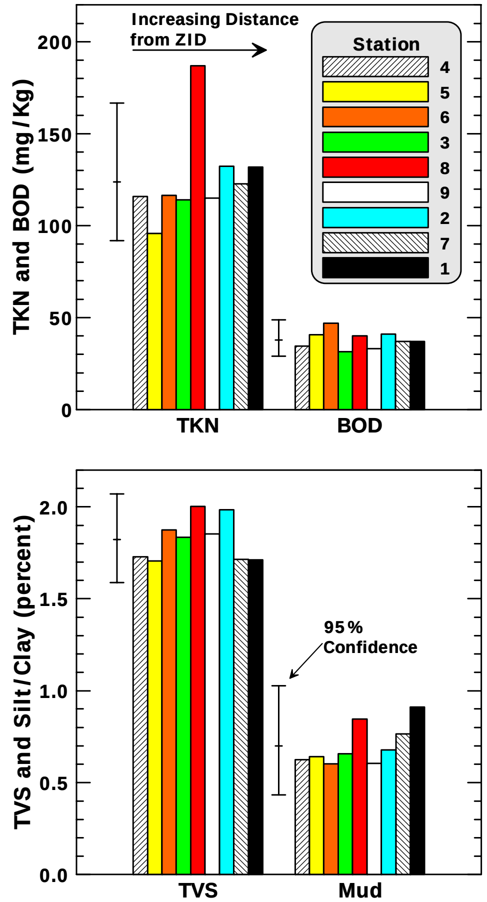

Figure 4.6 Organic Loading in Sediment Samples Collected during 2005

Temporal Trends Independent of Outfall Proximity

As suggested above, time histories of contaminant concentrations within sediments are a powerful tool for discerning the fate of contaminants within the marine environment. A buildup over time indicates that contaminants are accumulating within benthic sediments. Differences in the rate of buildup at various locations can reveal the cause of the contamination, particularly if it is related to a point-source discharge, such as a wastewater outfall.

Detailed analyses of sediment time histories have never revealed a buildup of contaminants related to the discharge of wastewater from the MBCSD outfall. This is largely due to the very low emission of contaminants discharged by the outfall. As described in Chapter 2, low contaminant emissions arise both because of a comparatively low discharge rate, and because contaminant concentrations within the effluent are low. Offshore sediment data collected during the October 2005 survey continue to demonstrate the lack of benthic impacts from MBCSD discharge. When added to historical data, time histories of sediment chemistry data now span two decades.

Figure 4.7 and Figure 4.8 show time histories for the six trace metals that have consistently been detected in Estero Bay sediments. Time histories for measurable organic constituents within sediments are shown in Figure 4.9. Concentrations measured in sediment samples collected at the two stations closest to the

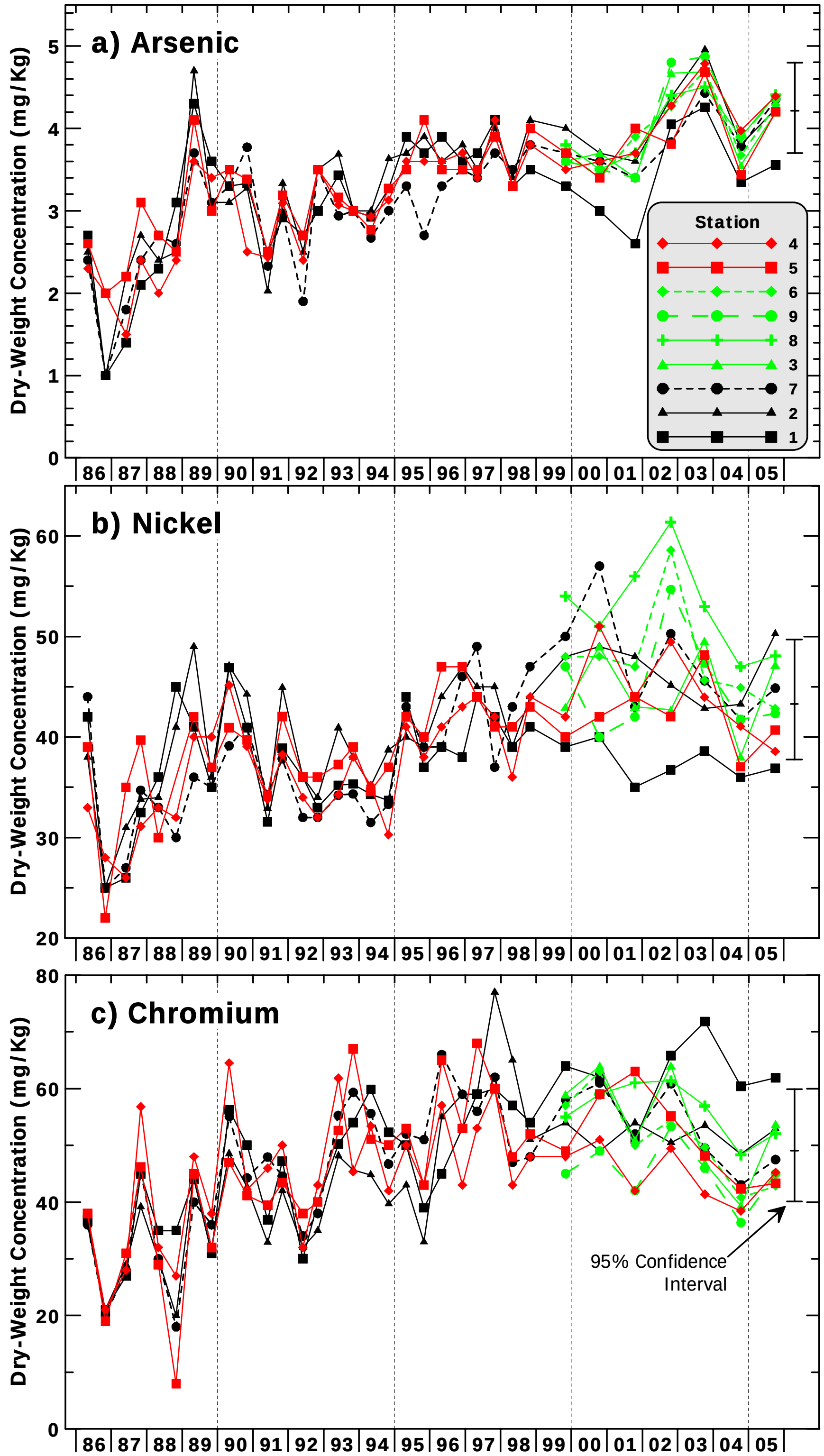

Figure 4.7 Records of Arsenic, Nickel, and Chromium Concentrations in Sediments

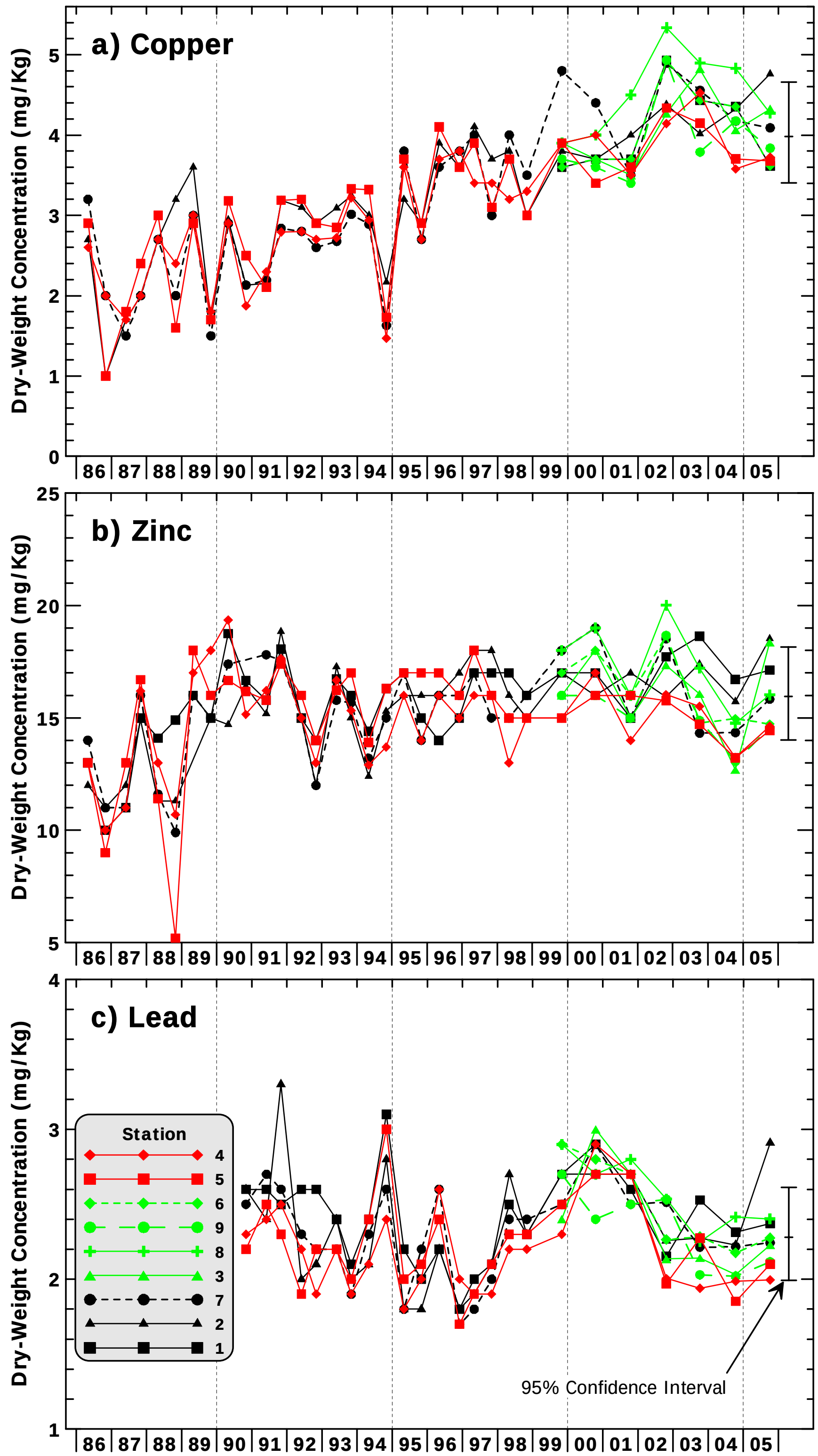

Figure 4.8 Records of Copper, Zinc, and Lead Concentrations in Sediments

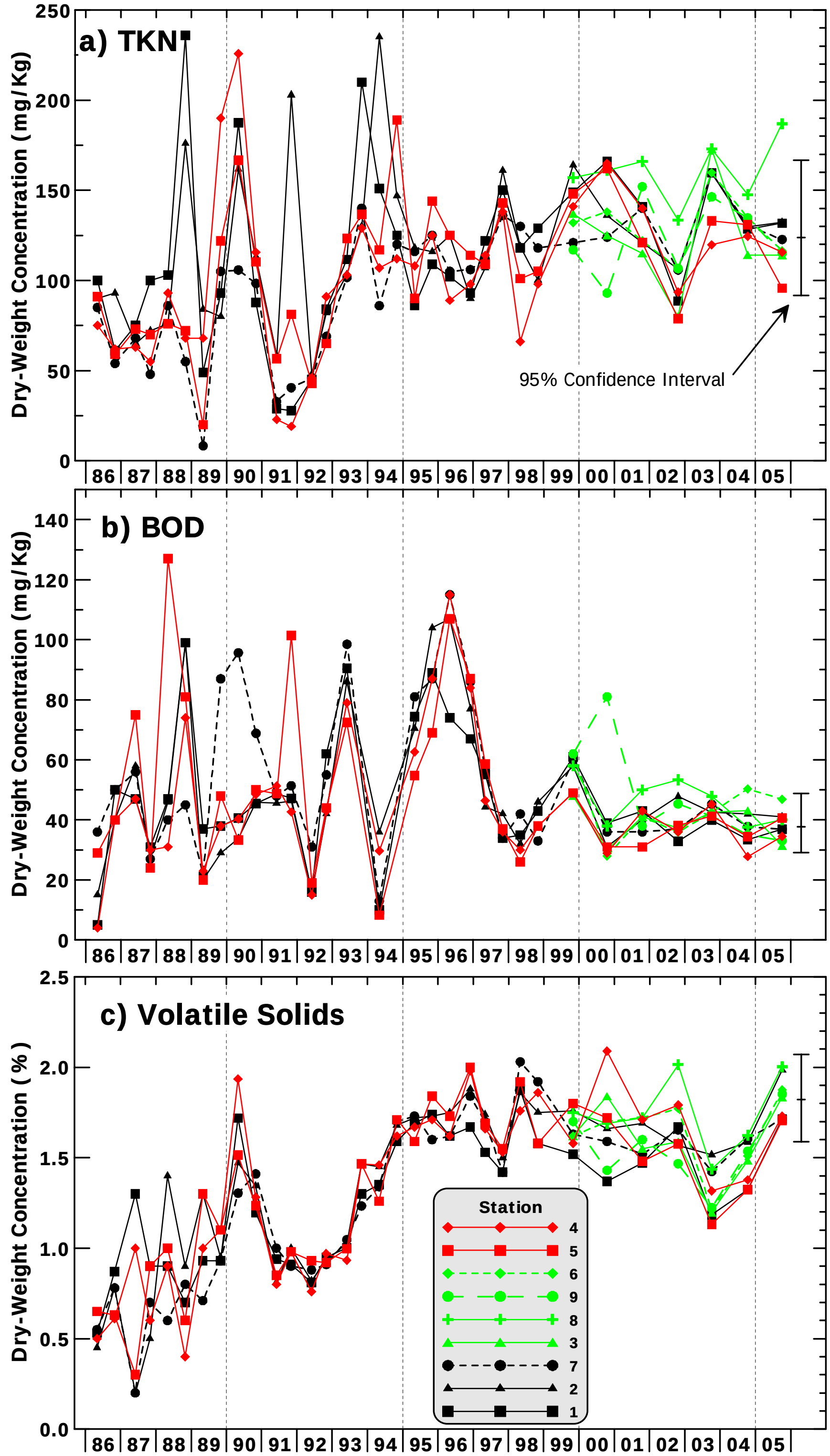

Figure 4.9 Records of TKN, BOD, and Volatile Solid Concentrations in Sediments

diffuser, namely ZID Stations 4 and 5, are indicated in red. Concentrations at nearfield stations (3, 6, 8, and 9), which are located approximately 60 m from the center of the diffuser, are indicated by green symbols. These stations were added to the monitoring program in 1999, so they currently have a relatively short time line of only seven years. Finally, concentrations at midfield stations (2 and 7), which are 150 m from the diffuser, and the upcoast reference station (1) are indicated in black.

A discharge-related accumulation of contaminants would be evident in these figures as a gradual separation of the colors over time. The red stations would eventually be the highest, followed by green, with black the lowest. However, this temporal signature of contaminant accumulation is not evident in any of the records of trace-metal or organic parameters. On occasion, some metals have exhibited a comparatively consistent offset for several years in a row, particularly at Stations 1 and 8. For the most part, however, the relative ranking of station concentrations fluctuated randomly from survey to survey. In fact, within any given survey, the range in concentrations among stations was almost always smaller than the variability among replicates collected at any given station. This within-station variability is indicated by the 95% confidence intervals shown to the right of each time history. It demonstrates that spatial differences at any given time were generally not statistically significant. In addition, interannual and seasonal fluctuations in concentration tend to be much larger than the spatial differences among stations that are observed at a particular time. Thus, the temporal trends and large fluctuations tend to occur in unison at all stations. This indicates that large-scale processes are influencing metal concentrations at all the monitoring stations together. This further suggests that changes in regional deposition and erosion affect northern Estero Bay as a whole, and that these changes overwhelmingly dominate fluctuations in the chemical properties of Estero Bay sediments.

Despite the overwhelming influence of regional processes, subtle differences in sediment properties at Stations 1 and 8 are occasionally apparent in the records of chemical concentrations. As with regional processes, these slight differences are due to an inherent variation in depositional processes rather than the influence of the MBCSD discharge. As described previously, Stations 1 and 8 are located in more quiescent depositional environments and consequently, tend to have a slightly higher fraction of fine sediments. As a result, slight differences in trace-metal concentrations can also occasionally be discerned at these stations.

For example, in the past five-to-seven years, a consistent offset in arsenic, chromium, and nickel concentrations measured at the reference Station 1 has become apparent. However, the anomalous arsenic and nickel concentrations appear to be regressing back toward mean conditions. In particular, the nickel offset at Station 1, which is delineated by the solid black line and black squares (■) in Figure 4.7b between 2001 and 2003, is less apparent in recent years because concentrations at other stations have dropped to levels similar to that of Station 1. A similar offset is evident in the markedly reduced arsenic concentrations at Station 1 between 1998 and 2001 (Figure 4.7a). However, in 2002, Station-1 arsenic concentrations returned to levels comparable to the average at other stations. Similarly, over the last three years, chromium concentrations at Station 1 appear to be consistently elevated compared to the other stations (Figure 4.7c). Elevated chromium concentrations at Station 1 are incongruous with comparatively depressed arsenic and nickel concentrations that occurred around the same time. This suggests that the depositional material accumulating at Station 1 differs from that of the other stations. Namely, the material being deposited at the reference station is enriched in arsenic and nickel relative to chromium.

Similarly, the records of nickel, copper, and TKN exhibit slight but consistently higher concentrations at Station 8 compared to most other stations. The time history of chemical concentrations in sediments collected at Station 8 are delineated by the solid green line with plus-sign symbols (+) in the figures. In contrast to the anomalous behavior at Station 1, the offset in concentrations at Station 8 has been relatively consistent across the six-year duration of sample collection at that station. Specifically, chemical concentrations in Station 8 samples were consistently among the highest measured for all parameters, but particularly in nickel (Figure 4.7b), copper (Figure 4.8a), and TKN (Figure 4.9a).

The higher concentration in sediment properties measured at Station 8 is probably related to the higher fraction of naturally occurring fine particulates present there. Station 8 is the only station that lies offshore of the diffuser structure in deeper water. Based on drifter measurements and plume distributions determined in the water-quality component of the monitoring program, the effluent plume is preferentially transported along the 15.2-m isobath and rarely extends offshore toward benthic Station 8 (see the current rose in Figure 7 on Page II-22 of MRS 2003a). In contrast, a naturally occurring offshore increase in background metal concentrations is often observed in benthic monitoring programs (Schiff and Gossett 1998, Steinhauer et al. 1994, Coats 1995). This concentration increase has been ascribed to the increase in fine particulates that naturally occurs with increasing depth and distance from shore.

The comparatively elevated iron and mud concentrations at Station 8 (Table 4.6, Figure 4.4, and Figure 4.1) further confirm the roll of natural processes. As discussed previously, iron and mud fractions are commonly used as conservative tracers to differentiate natural from anthropogenic components of variation in sediment properties (Dossis and Warren 1980, Horowitz and Elrick 1987). Offshore of southern California, Schiff and Weisberg (1997) found that iron was the optimal reference element for reducing natural variability in trace-metal distributions. At the MBCSD monitoring sites, normalization by iron removed much of the concentration differences observed at Station 8. This is apparent from a comparison of the chromium, nickel, zinc and copper bar charts in Figure 4.3 and Figure 4.5 on Page 4-22. The orange bar that represents the concentration at Station 8 is noticeably higher than other stations in the raw concentrations shown in Figure 4.3. Normalization by iron, shown in the bottom frame of Figure 4.5, results in substantially more uniform concentrations across all stations, largely because of reductions in the concentrations at Station 8.

Contaminant Accumulation Rates

This section quantifies long-term increases that are visually evident in some of the time histories shown above. Because the accumulation generally occurs at all stations, including stations distant from the outfall, the long-term increase is unrelated to effluent discharge. Table 4.9 presents the results of a series of linear regressions applied to the record of average contaminant concentration. The pair of columns labeled ‘*P*-Value’ reflects the statistical confidence in the rates determined in the next pair of columns. *P*-values less than 0.05 are considered significant and the associated rates, highlighted in bold, are considered reliable.

Two separate straight-line regressions were performed on the data for each parameter. The one labeled ‘*All*’ spanned the entire 20-year time series and included means from each of the 33 available surveys. A subset of only the recent, post-1990 data was analyzed separately. Visual inspection of the time series plots indicates that after 1990, the precision of chemical analyses appears to have improved. For many constituents, the scatter in measured concentrations within an individual post-1990 survey is smaller than in surveys prior to 1991. This difference is particularly evident in the time series of TKN, BOD, and TVS (Figure 4.9) where large outliers are less prevalent in recent years. In recognition of this temporal difference in data quality, trends were quantified using two data sets, one with all of the data, and another with only the data collected after 1990.

Table 4.9 Temporal Trends in Sediment Contaminant Concentration

	P-Value[a]		Rate[c] (mg/Kg/yr)		Effects[d]	Years to Effect[e]		Emission (Kg/yr)[f]		
Parameter	**Post-1990[b]**	**All**	**Post-1990**	**All**	**Level**	**Post-1990**	**All**	**Actual**	**Post-1990**	**All**
Arsenic	<0.001	<0.001	**0.10**	**0.10**	8.2	39	40	<4.2	9	9
Chromium	0.114	<0.001	0.57	**1.22**	81	—	18	<10.8	—	108
Copper	<0.001	<0.001	**0.12**	**0.12**	34	243	247	21.4	11	11
Nickel	<0.001	<0.001	**0.82**	**0.69**	51.6[g]	5	7	<13.3	72	61
Lead	0.869	—	0.00	—	46.7	—	—	—	—	—
Zinc	0.646	0.010	0.03	**0.16**	150	—	840	<91.5	—	14
TKN	0.008	0.002	**3.89**	**3.12**	—	—	—	—	—	—
BOD	0.728	0.943	-0.47	-0.06	—	—	—	—	—	—
TVS	0.003	<0.001	**0.05**	**0.06**	—	—	—	—	—	—

[a] Trend significant if *p*-value less than 0.05
[b] Regression performed on more-reliable data collected after 1990
[c] Rate of concentration increase per year
[d] Concentration where biological effects become evident (ERL)
[e] Predicted number of years before concentrations reach biological effects levels (from year 2005)
[f] Actual mass-emission in 2005 compared to that needed to account for annual concentration increase
[g] Effects-range median (ERM) for nickel because its concentration currently exceeds the ERL

Almost all detected contaminants exhibited a statistically significant ($p<0.05$) increase in concentration over time. The only exceptions were lead, zinc, and BOD. Lead has only been measured since October 1990 (Figure 4.8c). Although it does not exhibit a long-term trend over the entire record, its concentrations revealed a long-period fluctuation between October 1996 and October 2002. In contrast, zinc concentrations increased slightly between 1986 and 1991, but leveled off in recent years (Figure 4.8b). Consequently, although the zinc depositional rate determined from the entire 20-year time history was statistically significant ($p=0.010$), in recent years it was not ($p=0.646$). Long-term trends in BOD could not be reliably resolved in either time period because of wide interannual fluctuations that occasionally occur as a result of interferences that randomly appear in a few of the analyses (Figure 4.9b).

The relative importance of depositional rates for other sediment constituents is not obvious from a review of their magnitude alone. Table 4.9 indicates that chromium and TKN have the highest rates of concentration increase, with magnitudes exceeding 1 mg/Kg per annum. However, their initial concentrations were relatively high, so their percentage increase over time is similar to that of arsenic, copper, and TVS. In actuality, arsenic, chromium, copper, and TVS exhibited the largest increases relative to concentrations measured at the start of monitoring in 1986. The average copper concentration within sediments doubled in the past two decades (Figure 4.8a). Similarly, volatile-solid concentrations tripled since 1986 (Figure 4.9c). However, most of the TVS increase occurred between 1993 and 1995 in contrast to the copper increase, which resulted from a steady trend throughout the two-decade monitoring period. In addition, the TVS record shows that the precision of TVS determinations significantly improved after 1990, then declined again after 1998. Before 1990, laboratory analysis techniques were not as well refined as those of the subsequent analyses. The increased scatter after 1998 resulted from a switch to the analysis of a single grab sample. Sampling prior to 1998 consisted of an analysis of three grab samples that were either analyzed separately, or were composited and then analyzed. The replicate sampling conducted prior to 1998 lent additional stability to the reported concentrations for all sediment constituents.

Biological Implications

The relative importance of contaminant accumulation rates can also be evaluated in terms of their future potential for marine impacts. Estero Bay is one of the more pristine offshore regions along the western coast of the United States. However, continued accumulation of sediment contaminants at the current rates could degrade the marine environment to levels comparable to some of the more-industrialized regions within Monterey Bay and the Southern California Bight. As described in this section, if the increases continue, some trace-metal concentrations could eventually reach levels where adverse impacts to marine biota might be considered possible.

The biological implications of the observed contaminant accumulations are assessed in the '*Years to Effect*' columns of Table 4.9. They project the number of years that it will take for contaminant concentrations within Estero Bay sediments to reach toxicological benchmarks, assuming they continue to accumulate at historical rates. Four trace metals, arsenic, chromium, copper, and nickel, are of particular interest because they exhibit highly significant ($p \leq 0.001$) long-term increases in sediment concentration. The trends are visually evident in Figure 4.7a through Figure 4.8a, and, except for perhaps chromium, there is no evidence that concentrations have stabilized in recent years.

The '*Years to Effect*' characterizes accumulation rates for these four trace metals in terms of their marine biological importance. Copper accumulations are of least concern. Currently, sediment copper concentrations (4.0 mg/Kg) are far below the marine effects level (ERL=34 mg/Kg) and even if they continue to increase at the rate measured over the last decade, they will remain benign for two and a half centuries (247 years). Similarly, current arsenic concentrations (4.2 mg/Kg) are just over half of the lowest marine toxicological benchmark (ERL=8.2 mg/Kg). With annual increases of only 0.1 mg/Kg, it will be four decades before sediment arsenic concentrations might be expected to begin to impact marine biota.

Potential marine impacts resulting from future increases in chromium and nickel are of more immediate concern. Because their concentrations are already naturally high in the marine sediments of northern Estero Bay, small annual increases will cause contaminant levels to rapidly approach marine toxicological benchmarks. If chromium concentrations continue to increase at approximately 1.22 mg/Kg each year, contaminant levels could begin to affect marine organisms in as little as 18 years, or by the year 2021. Over the past two decades, nickel concentrations within the sediments of northern Estero Bay have consistently exceeded the ERL where marine impacts are normally thought to begin to be evident. However, as described in Section 4.2.1, confidence in the effects-levels for nickel are weak. Nevertheless, nickel concentrations are rapidly approaching the ERM where marine biological impacts are considered probable. At current accumulation rates, mean nickel concentrations are predicted to reach potentially toxic levels in as little as five years, by 2010.

However, these predictions are based on bulk metal concentrations rather than on dissolved metal concentrations. Dissolved concentrations are more indicative of potential biological impacts due to their increased bioavailability in the dissolved phase. Pore-water samples collected during the 2003 benthic survey contained average concentrations of arsenic (2.3 µg/L), chromium (0.7 µg/L), copper (0.4 µg/L) and nickel (1.5 µg/L) that were all far below the most-stringent saltwater criteria promulgated by the USEPA, which are 36 µg/L, 50 µg/L, 3.1 µg/L, and 8.2 µg/L. Moreover, the arsenic, chromium, and copper concentrations measured in the pore-water samples were comparable to background concentrations in clean coastal seawater. In contrast, the nickel concentrations that were measured in the pore-water samples were about three times higher than those observed in clean coastal seawater samples. However, the slightly elevated dissolved nickel concentrations were probably due to incomplete filtration of the pore-water samples. The presence of naturally occurring sediment particulates in the pore-water samples was apparent from the elevated dissolved-aluminum concentrations that were also found in the samples. In any regard, it is clear that the vast majority of the trace-metal concentrations measured in the bulk sediments arise from metal that is bound into the mineral matrix of sand grains, where it has little influence on marine organisms. Even if the dissolved nickel concentrations were slightly elevated within pore water, it is likely that indigenous populations would have adapted to the higher ambient concentrations that have probably existed for millennia.

Low MBCSD Metal Emission

The foregoing discussion identifies three trace metals, arsenic, chromium, and nickel, whose projected concentration increases could cause them to approach effects thresholds within a few decades. However, the increases in these metal concentrations are unrelated to wastewater discharge from the MBCSD outfall. As described in previous sections, the observed concentration increases are widespread and there are no consistent spatial gradients related to outfall proximity. Moreover, no heavy industry discharges into the MBCSD collection system, and trace-metal concentrations within effluent are uniformly low; so low in fact, that MBCSD metal emissions do not appreciably contribute to the accretion of these metals in marine sediments. This is demonstrated with an elementary mass-balance computation.

The right-hand columns of Table 4.9 compare the actual year-2005 mass-emission of each trace metal (from Chapter 2) with the lowest conceivable emission that could cause the concentration increase observed within surficial sediments surrounding the outfall. For all three metals of concern, actual mass emissions from the MBCSD outfall during 2005 were well below the emission that would be required to achieve the observed annual increase in sediment concentrations. Specifically, the 4.2-Kg upper-bound arsenic emission was less than half of the 9-Kg annual emission that would be required to raise sediment arsenic concentrations by the observed 0.1 mg/Kg each year. Similarly, an upper bound mass emission of less than 10.8 Kg of chromium cannot account for the observed 1.22-mg/Kg/yr increase in sediment chromium concentrations. In fact, the emission would have to be at least ten times higher. Finally, the annual increase observed in nickel concentrations could be explained only by a discharge of well more than 61 Kg, while the actual discharge in 2005 was probably far less than 13.3 Kg.

In reality, the differences are much larger given the conservative assumptions incorporated in the mass-balance computation. First, the '*actual*' mass emissions from the outfall represent upper bounds based on the effluent detection limit for the metals. Arsenic was the only one of these three metals that had a quantifiable concentration within the effluent (Table 2.8) and, consequently, the actual year-2005 emissions were probably far less than those listed in Table 4.9, which were largely computed from the PQLs of the chemical analyses. Second, the projected emission rates, namely those required to obtain the observed increase in sediment concentrations, were grossly underestimated. Metal concentrations are probably increasing within a much larger volume of seafloor sediments than was assumed in the mass-balance computation. The calculation conservatively assumed that the metals were accumulating only within the upper 2 cm of seafloor sediments where subsamples were collected in the field for subsequent chemical analysis. Moreover, the impact footprint was assumed to cover a radius of approximately 1 km, which represents the distance between the diffuser and the outermost sampling station. In reality, the concentration increases are probably occurring over a much wider area, and over a greater depth than that defined by the sampling limits of the monitoring program.

In any regard, the analysis of temporal trends in sediment contaminants demonstrates that contaminants are not preferentially accumulating near the outfall. In addition, upper-bound estimates of emissions from the outfall are too small to have materially contributed to the observed accumulation of trace metals within the sediments of northern Estero Bay.

4.3 BENTHIC INFAUNAL COMMUNITY

The infauna living within the surficial sediments of northern Estero Bay act as sentinels for possible impacts caused by the discharge of MBCSD wastewater. They serve as indicators of marine pollution because they have limited mobility and cannot easily escape exposure to contaminants in their immediate environment. Also, some species are more sensitive to pollutant stress than others, and changes in relative abundance can imply degraded environmental conditions. Because infauna reside within seafloor sediments, they are closer to the potential source of pollution. The diffuser structure lies on the seafloor, where particulate contaminants discharged into marine waters ultimately settle and accumulate. Infauna are also important organisms to monitor because of their low trophic level within the marine food chain. They are a major food source for more-mobile epifaunal and pelagic marine organisms, such as crabs, finfish, and marine mammals. Finally, many infauna are filter feeders that may bioaccumulate contaminants even when standard chemical assays of water samples are unable to detect low-level contamination.

In recognition of infauna's role as an early detector of marine pollution, the MBCSD discharge permit requires regular monitoring of the overall health of the benthic community within Estero Bay. Infaunal monitoring also addresses the COP requirement that the discharge not degrade indigenous marine biota. Benthic monitoring has now accumulated two decades of infaunal data. During that time, 142,000 specimens representing more than 347 individual taxa have been examined. Throughout the monitoring program, there has never been any indication of deleterious outfall-related impacts to the benthic community. Instead, the data have revealed a consistently healthy indigenous infaunal community with a uniformly high diversity that does not decrease with proximity to the diffuser.

These observations hold true despite widespread variation in the abundance of individual organisms that have occurred throughout the monitoring program. Over the years, the dominance of individual taxa have waxed and waned in response to natural changes in the environment. For many taxa, distinct differences are evident between the winter and summer sampling seasons. Others come and go in response to powerful long-term variations in the oceanographic environment. One such interannual variation is the El Niño – Southern Oscillation, whose impact was evident in benthic sediment samples collected within Estero Bay on at least three occasions during past the two decades of MBCSD monitoring. The first event was in 1988, and the second began at the end of 1997 and extended into 1998. Its influence on benthic organisms was still evident during 1999. Another minor event may have occurred in 1991.

Over 4,700 individual organisms were enumerated in the 45 sediment samples that were collected at nine stations during the benthic survey conducted in October 2005. A compendium of the taxonomic identifications and enumerations conducted on year-2005 sediment samples is provided in Tables E.1 through E.10 at the back of this report. Table E.11 and Table E.12 of Appendix E list infaunal indices that were computed from infaunal data, as required by the NPDES monitoring and reporting program. This section of the annual report combines these data with biological data collected over the prior 19 years to determine whether there have been discharge-related impacts to the benthic biota.

Three analyses demonstrate that there have been no deleterious impacts to benthic organisms as a result of the discharge. First, the organisms that inhabit the sediments around the outfall are indicative of an indigenous community of filter feeders that thrive only in clean sediments. Second, infaunal data collected during 2005 show no evidence of spatial gradients indicative of biotic degradation attributable to effluent discharge. Third, despite wide fluctuations in community parameters over time, there were no long-term trends indicative of an increasingly degraded benthic environment around the outfall. All the stations tended to respond in unison to large interannual and seasonal influences on the infaunal community. To date, the indigenous benthic community remains healthy, with no perceptible influence from ambient contaminant levels, including the previously discussed long-term increases in trace-metal concentrations that have occurred throughout Estero Bay.

4.3.1 A Healthy Indigenous Infaunal Community throughout Estero Bay

The type and abundance of benthic organisms directly reflect the quality of the marine sediments where they live. Degraded sediments have quantifiable differences in the composition of the biological community that lives on and within them. They tend to have low richness and diversity because they are populated by a few opportunistic pollution-tolerant taxa that feed on detritus and thrive in sediments that are high in subsurface organics. In contrast, clean sediments are populated by a diverse assemblage of organisms that include filter feeders, which extract nutrients from suspended particulates. These suspension feeders are sensitive to excessive particulate loads such as those discharged from treatment works. The widespread presence of suspension feeders around the MBCSD outfall clearly indicate that the discharge is not adversely affecting them. One such suspension feeder is the club-tipped anemone (*C. californica*). It is found carpeting the diffuser ports, as depicted in the photograph on Page 2-5. Its presence is indicative of the benign nature of the suspended sediments discharged by the treatment plant.

Excluded Organisms

The populations of two organisms, *Diopatra ornata* and *Dendraster exentricus,* have a profound effect on the MBCSD infaunal database. However, these organisms and their attached taxa are not particularly sensitive to changes in sediment quality, and their presence confounds impact assessments. Consequently, their populations were removed from the database prior to the impact assessments described in this section.

Since 2000, the parchment tubes of the Ornate Tubeworm (*Diopatra ornata*) have been separated in the field prior to identifying and enumerating the infauna. However, in contrast to surveys conducted since that time, there were no adult *D. ornata* tubes found during the October 2005 survey. Removal of the parchment tubes in the field, prior to sieving, avoids the inclusion of large populations of epiphytes that live among the pieces of shells, algae, sticks, and other debris that are attached to these large tubes (Figure 4.10). Including large numbers of nestling epifauna that reside on the tube casings, particularly small crabs and other crustaceans, skews the enumeration of marine organisms that are truly infaunal in nature. Because epifauna live on the surface of marine substrates, they tend to be more mobile and less susceptible to sediment pollutants than the infauna, which live in the interstices between sand grains. The tubeworms themselves, however, were included in the enumerations because they tend to congregate in higher densities close to both natural reefs and man-made structures, such as outfalls (Davis et al. 1982). As discussed below, the absence

Figure 4.10 Photograph of a Tubeworm (*Diopatra*)

of *D. ornata* tubes in 2005 contrasts with populations found in prior years and bears testimony to the large size of interannual variations that occur in the benthic community.

Statistical analyses also excluded Pacific Sand Dollar (*Dendraster excentricus*) populations. Sand Dollars were removed from the database prior to statistical analysis because they exhibit extreme population fluctuations from year to year (Figure 4.11). Because their large numbers overwhelmingly dominate the population statistics during high-recruitment years, they have the potential to mask any influence of the discharge on less-dominant taxa whose populations are more consistent and sensitive to potential discharge impacts. For example, during the October 2002 survey, Sand Dollars were by far the most abundant organism collected. In contrast, the Sand Dollar population in the following year (2003) was exceptionally low, and the removal of the 87 specimens of *D. excentricus* from the database had little impact on the statistical analyses. Similarly, in 2005, Sand Dollar population was again low. Consequently, community indices computed both with (Table E.11) and without (Table E.12) these 168 specimens were similar. The difference in station mean indices was generally much smaller than the standard deviation among replicate samples. However, in other years, inclusion of *D. exentricus* unduly influenced the statistical analyses, making interannual comparisons difficult.

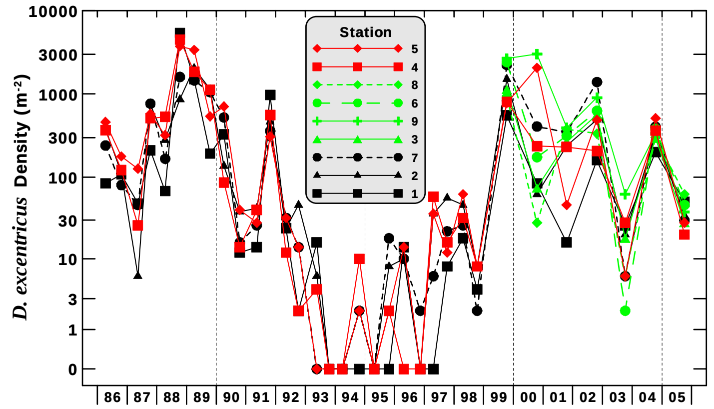

Figure 4.11 Time History of Pacific Sand Dollar (*D. excentricus*) Density

Within the monitoring area, the distribution of the transient Sand Dollar population was unrelated to effluent discharge. As with the chemical constituent time histories, spatial distributions of Sand Dollar populations related to outfall proximity would be evident in Figure 4.11 as a separation between the time histories for the ZID stations (red), for midfield stations (green), and stations well away from the diffuser structure (black). Instead, large fluctuations in Sand Dollar abundance tend to occur equally at all stations. The consistently low density of fully grown adult Sand Dollars, even during the high-population periods, demonstrates that their presence is transient and that there are no permanent Sand Dollar beds in the region (Brown and Caldwell 1989, 1993). It is well established that Sand Dollar beds are often dominated by a single age group (cohort) suggesting that recruitment is episodic and only occasionally successful (Cameron 1980, Oliver et al. 1980, Morin et al. 1985, Cameron and Rumrill 1982).

Although Sand Dollars are filter feeders and can be sensitive to pollution, external environmental effects cause their populations to fluctuate within northern Estero Bay. Specifically, the increased Sand Dollar recruitment that occurred in late 1988, 1991, and 1999 was probably associated with high-waves generated by El Niño events in 1988, 1998, and the weaker one in 1991. Sand Dollars form dense beds just seaward of the wave break, with disks on edge protruding at an angle from the sand and aboral sides facing into the current. The seaward border of the bed ends abruptly close to the wave break, which acts as a barrier for predacious sea stars. As a result, predation is limited because the sea stars can move over the bed only at significant risk of tumbling into the surf zone with a passing wave (Morin et al. 1985). Consequently, Sand Dollar beds migrate shoreward in the summer and seaward in the winter in response to seasonal wave action (Oliver et al. 1980). A winter of strong wave action, usually associated with El

Niño, reduces the numbers of predators, permitting recruitment of juvenile Sand Dollars. On numerous occasions during the 2002 survey, Short Spined Sea Stars (*Pisaster brevispinus*) became jammed in the jaws of the grab sampler. This caused the loss of the sediment samples and resulted in an unusually high number of repeated grab sampling attempts at several stations. Interestingly, Sand Dollar populations in 2003 were markedly diminished from those found in the 2002 survey, suggesting that predation may have played a significant role in the reduction of Sand Dollar populations between 2002 and 2003. The resurgence of a juvenile Sand Dollar population in the October 2004 survey indicates that 2004 represented a new recruitment year for Sand Dollars within the monitoring area.

Abundance

In this report, the overall health of the benthic community within Estero Bay is reflected by the principal population parameters computed from samples collected at each station over the duration of the monitoring program. For the reasons described above, Sand Dollars (*D. excentricus*) were excluded from the population statistics to better evaluate the overall health of the benthic community. The community parameters provide a succinct description of the abundance, diversity, richness, and trophic (feeding) structure of benthic biota. Figure 4.12 shows the temporal variability in mean community parameters that were computed for each of the nine benthic monitoring stations. Each major category of community parameter is discussed in the sections that follow.

Organism density, as shown in Figure 4.12a, measures the total abundance, or number of organisms, within a 1-m^2 area of the seafloor. It is the most basic single measure of a community of organisms, but lacks valuable insight as to how the abundance is distributed among the differing taxa. Because of this deficiency, it is not a useful ecological measure by itself. Nonetheless, major declines in organism density can occur after a rapid increase in sediment contamination. This initial decline is often followed by a marked increase in total abundance when opportunistic species repopulate the degraded environment.

Throughout the monitoring program, the number of organisms collected at each station has fluctuated enormously (Figure 4.12a). Although the range in abundance covers almost an order of magnitude, differences in abundance among individual stations within any given survey tended to be much smaller. Namely, all the stations tend to track the temporal fluctuations as a group, and spatial differences among stations within a given survey were generally small compared to the wide temporal fluctuations between surveys.

The largest population changes occur on a seasonal basis. Post-summer populations are generally a factor of three higher than the post-winter populations. The reduced winter population, which was only measured prior to 1999, is a response to harsher oceanographic conditions. Winter storms generate intense wave-induced currents that scour the bottom near the 15-m water depth of the outfall. Summer populations reflect recruitment of the more delicate annelid worms that can only survive in more quiescent benthic-flow regimes.

Any spatial gradients are completely masked by the large temporal fluctuations. In particular, the abundance at stations near the outfall (shown in red) was not consistently lower (or higher) than the density at distant stations (shown in black) throughout the monitoring program. This demonstrates that any accumulation of sediment contaminants from the effluent discharge over the past two decades has had no discernable impact on infaunal abundance, particularly when compared to large seasonal and interannual population fluctuations.

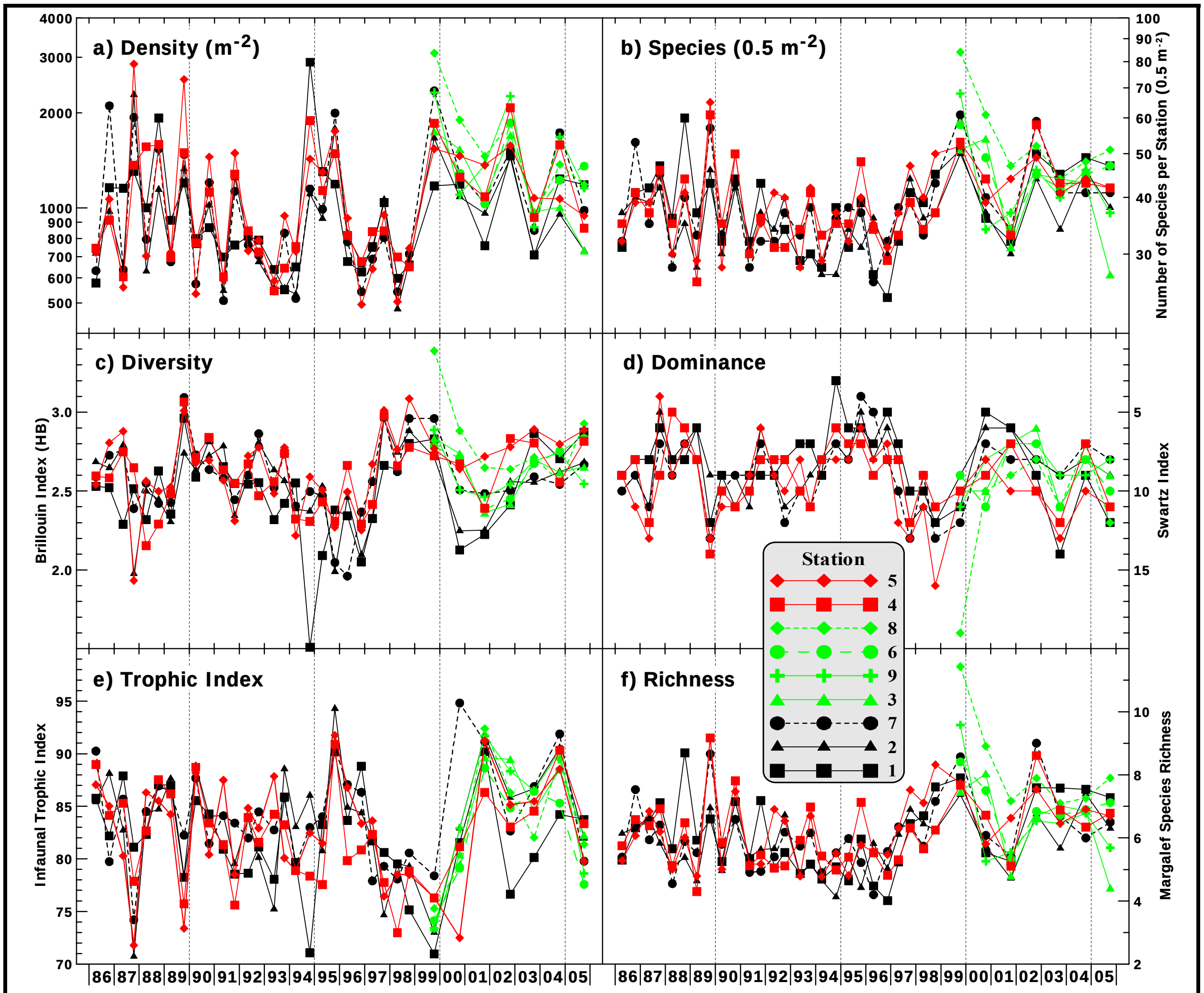

Figure 4.12 Time History of Infaunal Community Parameters

Number of Species

The total number of species is another measure of the overall health of benthic communities. Everything else being equal, infaunal communities can respond to environmental stresses by becoming less diverse, as reflected by a substantial reduction in the number of species. However, the number of species encountered in a sample is directly related to the number of organisms in the sample, i.e., it is inherently related to sample abundance. Because rare species are generally undersampled, the number of species automatically decreases in samples with fewer organisms. Thus, it is difficult to tell if a decreased number of species is due to an actual decline in biodiversity, or simply due to decreased sample sizes (abundance). Irrespective of the influence of sample size, reductions in species counts would suggest a decline in diversity and degradation in the marine environment.

There was no obvious temporal trend in numbers of species that would suggest that sedimentary conditions near the outfall were being preferentially degraded (Figure 4.12b). Specifically, there was no divergence in the colors over time, where species counts at stations distant from the outfall, shown in black, began increasing relative to stations near the outfall. On the contrary, the largest numbers of species were often observed at the stations closest to the outfall diffuser (Stations 4 and 5, shown in red). However, as with abundance, these spatial differences were minor compared to seasonal and interannual fluctuations. Beginning in 1996, there was a steady increase in the number of species encountered in samples. This trend culminated in 1999, when 84 species were collected at Station 8 alone. However, this increase was largely due to the increased prevalence of Ornate Tubeworms (*D. ornata*) within some of the samples. Before that time, between 1990 and 1996, at most one, but usually no *D. ornata* specimens were collected during any given survey. In the post-winter surveys of both 1997 and 1998, two Ornate Tubeworms were encountered. By 1999, four of these tubeworms were found at Station 8 alone. These tubes were not removed prior to field sieving, and the elevated presence of epiphytic taxa resulted in the unusually large number of species recorded in 1999.

The other temporal fluctuations in species counts tended to track the fluctuations in abundance. Consequently, at least some of the variability in the numbers of species was an artifact of the larger sample size (abundance) rather than purely an increase in species richness or diversity. A case in point is the comparatively low number of species found at Station 3 in 2005 as shown by the rightmost green triangle (—▲) in Figure 4.12b. The 27 species found at that station was perceptibly fewer than the 43 species found at most other stations in 2005. However, as shown in Figure 4.12a, the abundance at Station 3 was also low compared to the other stations in 2005. This is a clear demonstration of the inherent relationship between sample size (abundance) and species counts.

Diversity

A large number of interrelated diversity indices have been developed in an attempt to account for the inherent relationship between the number of species and sample size or abundance. Two common indices are the Shannon Diversity (*H'*) and the Brillouin index (*HB*) (Appendix E.2). Both exhibit nearly identical time histories in the MBCSD database. Consequently, only the Brillouin Index is shown in Figure 4.12c. As expected, the unusual nature of the species counts at Station 3 in 2005 is not reflected in the diversity index because it accounts for the reduced abundance at that station.

Ostensibly, diversity should decrease in a degraded benthos, although significant reductions can also occur in the absence of anthropogenic stresses. Nevertheless, temporal trends in diversity did not exhibit any evidence of a deteriorating benthic environment near the outfall. On the contrary, in recent years the highest diversity was typically observed at one of the two stations located closest to the outfall (shown in red). Higher diversity at the two ZID stations indicates a healthier infaunal community close to the outfall rather than a community impaired by the discharge of effluent. Although not immediately apparent in the time series shown in Figure 4.12, diversity along with dominance, were the only two indices that exhibited statistically significant spatiotemporal trends. These trends are addressed through tests for parallelism that are presented in Section 4.2.3.

Diversity measured in recent years at nearfield Station 8, shown by green dashed lines connecting green diamonds (- - -◆- - -) in Figure 4.12c, warrants some discussion. Sampling at the nearfield stations started in 1999, as shown by the green time lines. As was the case for other indices, the large diversity outlier of 3.39 at Station 8 in 1999 was partially an artifact of the enumeration of additional epiphytes residing on the casings of the Ornate Tubeworms. However, following 1999, the infaunal community at Station 8 continued to exhibit anomalous properties despite removal of the parchment tubes in the field prior to sieving. Namely, in four of the seven years since sampling at Station 8 began, its community has exhibited some of the highest densities, species counts, diversity, and richness of any of the stations. In

2005, community indices at Station 8 were also slightly elevated, and as will be discussed in the multivariate analysis section (4.3.4 on Page 4-46), Station 8 supported an infaunal community distinct from that of other stations. Differences in the community structure at Station 8 can be ascribed to its location farther offshore, in slightly deeper water (see Figure 4.1 on Page 4-2). Because of its location, it is less affected by seafloor turbulence generated by shoaling waves as they impinge on Atascadero Beach. Ornate Tubeworms (*D. ornate*) and other delicate taxa are better suited to this more-quiescent environment, and this probably accounts for their increased abundance there.

Another historical outlier was the very low diversity (1.51) found at Reference Station 1 during the October 1994 survey (black square ■ in Figure 4.12c). This low diversity resulted from the greatly increased abundance of the spioniform polychaete worm *Magelona sacculata*. Nearly 1,000 specimens were collected at that one station alone during the October 1994 survey. However, its presence at that station was erratic, with more than 75% of the specimens residing in only two of the five replicate samples collected at the station. Magelonids are common in sandy bottom sediments, although their distribution is patchy, and the presence of this surface detritus feeder can be considered marginally indicative of degraded sediment conditions. In northern Estero Bay, it is often one of the numerically dominant taxa, but only approximately 50 specimens are typically collected at each station. Regardless, its abnormally increased abundance in 1994 occurred at the reference station, and bore no relationship to effluent discharge from the outfall.

Dominance

The Swartz species dominance differs from the other community indices because it does not make *a priori* assumptions concerning the distribution of individuals among species. Because it is non-parametric, it is less sensitive to the limitations that plague other community indices. It is defined as the total number of species accounting for 75% of the individual organisms collected. As such, it represents an inverted measure, in which increased dominance is associated with lower values of the Swartz index. Increased dominance is expected in degraded benthic communities where a few opportunistic, pollution-tolerant taxa abound in the absence of a wider array of pollution-sensitive organisms. As with the other community parameters, Figure 4.12d shows that differences in dominance among stations in a given survey were generally small compared to temporal fluctuations. Overall dominance was generally low, with five or more species typically sharing numerical superiority. In fact, in most surveys, organisms were more evenly distributed among taxa at the outfall stations (shown in red) than at more distant stations. This is opposite of the pattern expected if effluent discharge had degraded the benthic environment.

By far the lowest historical dominance occurred in samples collected at Station 8 during 1999. During that survey, nineteen species accounted for 75% of the specimens collected. Judging from a Swartz index that normally is near eight species, up to 10 of the 19 dominant species that were found at Station 8 during 1999 were epiphytic organisms associated with the presence of Ornate Tubeworms. The exclusion of these tubeworms from subsequent surveys has resulted in more representative Swartz indices.

Clean-Sediment Feeding Guilds

The Infaunal Trophic Index (*ITI*) (Word 1978 and 1980) compares the abundance of four soft-bottom assemblages that are distinguished by their feeding behavior. Because sensitivity or tolerance to organic enrichment differs among the four groups, shifts in the dominance of the individual feeding groups, as reflected in a changing Trophic Index, can be indicative of degraded environmental conditions. The *ITI* ranges between 0 and 100. When species in Group I (suspension feeders) and Group II (surface-detritus feeders) dominate, index values are above 58 and sediments are considered relatively clean. Lower infaunal indices occur when species in Group III (surface deposit feeders) and Group IV (sub-surface detritus feeders) dominate in sediments that are high in organic contaminants.

The *ITI* has been consistently high throughout the monitoring program, and the 2005 survey was comparable to trophic indices recorded in past surveys (Figure 4.12e). The *ITI* has always been well above the critical *ITI* of 58 that normally indicates degraded conditions. In fact, trophic indices have always exceeded 70, which confirms the consistent dominance of suspension and surface-detritus feeders in the relatively clean sedimentary environment of northern Estero Bay. Moreover, there is no pattern of degradation close to the outfall evident in the *ITI* time series. Effluent-related degradation would appear as a reduced *ITI* at the ZID Stations (4 and 5). Effluent-related degradation would also emerge in Figure 4.12e as a gradual separation in colors over time, with stations closest to the diffuser structure (shown in red) eventually having the lowest *ITI*. This would be followed by a slightly higher *ITI* at nearfield stations, shown in green. Instead, there is no consistent pattern of color separation from year to year.

The high *ITI*s measured in samples collected during 2005 reflect a healthy community where suspension feeders predominate. Specifically, the high trophic indices were a consequence of an infaunal population consisting largely of phoxocephalid amphipods, namely *Majoxiphalus major, Foxiphalus xiximeus*, *Rhepoxynius menziesi*, and *Rhepoxynius abronius* (see Table E.1). Together, these amphipods accounted for 20% of the non-*Dendraster* specimens collected during the survey. Phoxocephalids are active suspension feeders whose presence weighs heavily in the computation of the *ITI* as members of the Group-I feeding guild (See Page E-12 in the Appendix). The amphipod *R. abronius* was the third-most abundant taxon collected during the 2005 survey. It is also known to be particularly sensitive to pollution (Pearson and Rosenberg 1978) and, because of this, is often recommended for bioassay testing of marine sediments (ASTM 1992; USEPA-USACOE 1991, 1993). Its prevalence in samples collected at all stations in October 2005 lends confidence in the general lack of potential discharge-related impacts. This taxon is also particularly sensitive to increased amounts of fine sediments, and variations in grain-size distribution can often account for variations in its abundance (DeWitt et al. 1988). Not surprisingly, its greatest abundance was observed at Station 1 (94 specimens, Table E.1) and Station 8 (85 specimens) where mud fractions were elevated compared to the remaining stations (Figure 4.6 on Page 4-26).

In addition to substantial numbers of phoxocephalid amphipods, another pollution-sensitive taxon, the gammerid amphipod *Eohaustorius sincillus* was prevalent at all stations and accounted for nearly 14% of the non-*Dendraster* specimens collected overall (Table E.1). In contrast, opportunistic pollution-tolerant species, such as *Polydora ligni*, *Capitella capitata*, *Paraprionospio pinnata*, and oligocheate worms, were completely absent in samples collected during 2005. The prevalence of sensitive filter-feeding taxa and the general absence of pollution-tolerant opportunistic taxa led to the overall high levels of the infaunal trophic index in 2005. The uniformly healthy population found in benthic samples collected in 2005 was consistent with the infaunal population found in samples collected in the prior 19 years.

4.3.2 Spatial Distribution Unrelated to Infaunal Degradation from Effluent Discharge

As described in Appendix E.2, four of the biological indices are more informative than the other indices. They include the Brillouin Diversity Index, the Swartz Dominance Index, the Margalef Richness Index, and the Infaunal Trophic Index (*ITI*). The spatial distribution of average values for these four indices is shown in Figure 4.13. The indices were computed without *D. excentricus*, but because of only moderate Sand Dollar populations in 2005, their exclusion did not significantly affect the determination of the indices. Moreover, the Sand Dollar exclusion had no affect on the Infaunal Trophic Index, because *D. excentricus* is not one of the species included in the trophic (feeding) guilds incorporated in the index.

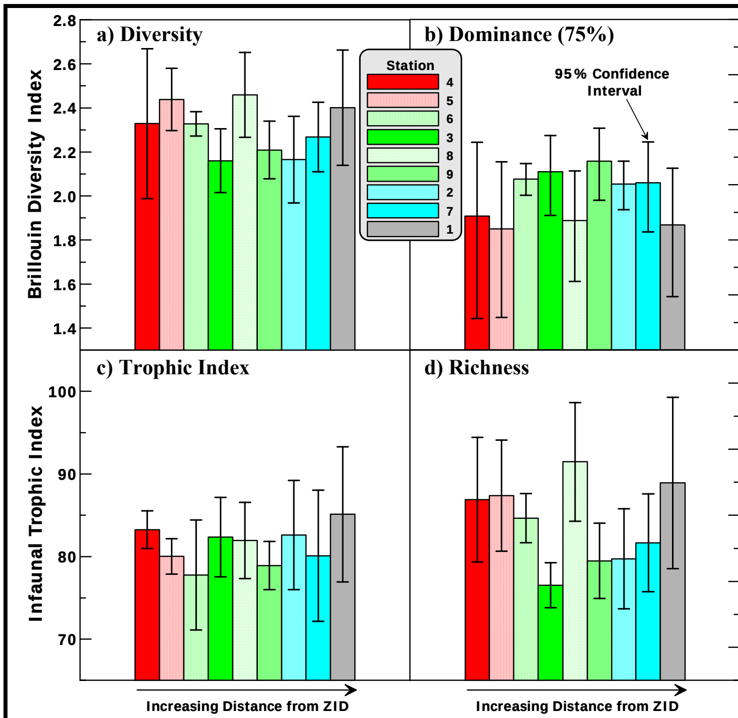

Figure 4.13 Distribution of Mean Biological Indices in Samples Collected during 2005

The bars shown in Figure 4.13 are arranged in order of increasing distance from the diffuser structure. The red-colored bars on the left are associated with the ZID stations located closest to the diffuser. The four green bars on the immediate right of the red bars represent the nearfield stations that were sampled for the first time in 1999. Finally, the blue and gray bars that are farthest to the right represent three distant stations, which lie 129 m or more from the outfall.

The height of the bars reflects the average index computed from the five replicate samples collected at each station. The 95% confidence intervals represent the within-station variability among replicates. To the extent that standard assumptions of normality and independence apply, there is a 95% chance that the confidence interval encompasses the true mean value. Thus, comparing means and confidence intervals between two stations is equivalent to a test of the null hypothesis that there is no significant difference in the means. Specifically, if the mean at one station is encompassed within the confidence interval at another station (or vice versa), then there is no reason to reject the null hypothesis at the 95% confidence level ($\alpha \leq 0.05$).

Visual inspection of Figure 4.13 indicates that some of the stations have mean indices that differ significantly from the means at other stations. A more rigorous analysis of variance (ANOVA) confirms that this is the case for diversity (p=0.025) and richness (p=0.0005), but not dominance (p=0.08) and *ITI* (p=0.27). After accounting for multiple hypothesis tests, the only significant difference in mean diversity was between Stations 3 and 8. Similarly, the mean richness at Station 8 was significantly higher than the richness at Stations 2, 3, and 9. Also, the mean richness at Station 1 was higher than that of Station 3. All other differences were not statistically significant at the 95% confidence level. As discussed previously in this chapter, Stations 1 and 8 have a greater amount of fine sediments due to their more quiescent locations. Increased muds and associated organic compounds probably account for the slight increases in the diversity and richness of the benthic community that resides in the surficial sediments at those Stations.

In any regard, the differences among stations are not consistent with a spatial pattern indicative of impairment from effluent discharge. As with the physicochemistry bar charts, spatial regression analyses were performed to determine whether any of the infaunal parameters exhibited a significant log-linear spatial trend. These regression analyses provide a more powerful test for gradients than the null hypothesis tests described above because they examine all the stations simultaneously. All seven of the community indices listed in the discharge permit and collated in Table E.12 were examined for spatial gradients extending from the diffuser structure. None of the indices had statistically significant gradients related to outfall proximity ($p \geq 0.3$).

4.3.3 Spatiotemporal Trends Unrelated to Infaunal Degradation from Effluent Discharge

As with the chemical constituents within northern Estero Bay sediments, subtle spatiotemporal trends in the infaunal community are difficult to discern in the time series shown in Figure 4.12 on Page 4-39. The parameters exhibit wide fluctuations between seasons and years. Compared to these large temporal oscillations, the spatial differences among samples collected at a given time (within a given survey) were comparatively small. In other words, the stations generally tracked one another through time. This indicates that changes in infaunal communities that arise because of the influence of regional processes within Estero Bay are far larger than those induced by localized effects from, for example, effluent discharge. By the same token, within a given survey, perceived spatial differences in infaunal parameters cannot be unequivocally ascribed to impacts from the discharge. Discharge-induced spatial differences in infaunal communities are confounded by larger natural differences in the sedimentary environment and other zoogeographic influences.

The best way to discern discharge-related impacts is from spatial differences that change over time. These discharge-related spatiotemporal changes would appear in Figure 4.12 as a gradual separation of the colors as contaminants accumulate near the outfall and affect the organisms there. However, because of the large coherent fluctuations that occur over time, subtle spatiotemporal changes may be present but not visually apparent in the time series. Fortunately, rigorous statistical tests that can definitively determine the presence of spatiotemporal change are available. These parallelism tests assess temporal trends in the difference between average indices at stations located near the outfall, and those that are distant from the outfall. Taking the difference in average values removes the coherent temporal oscillations that occur on a seasonal and interannual basis. In effect, the large temporal excursions that occur in unison at all stations are assumed to occur in response to regional influences that strongly affect benthic communities at all stations at once. This natural background variation is not pertinent to the impact assessment, and is effectively removed by examining the difference in long-term trends. These time series of differences in parameters at ZID and distant stations are presented in Figure 4.14.

If the discharge was impacting the benthic community over time, then the infaunal properties of populations at stations near the outfall (4 and 5) will slowly diverge from that of the distant stations (1, 2, and 7). This will appear as a trend or slope in the average difference shown in Figure 4.14. A regression test for a statistically significant trend (slope) in the difference is equivalent to a test of the null hypothesis that the population responses at impacted and non-impacted sites are parallel over time. Use of these parallelism tests successfully quantified the recovery of intertidal communities impacted by the *Exxon Valdez* oil spill in Alaska, when other techniques fell short (Coats et al. 1999, Skalski et al. 2001).

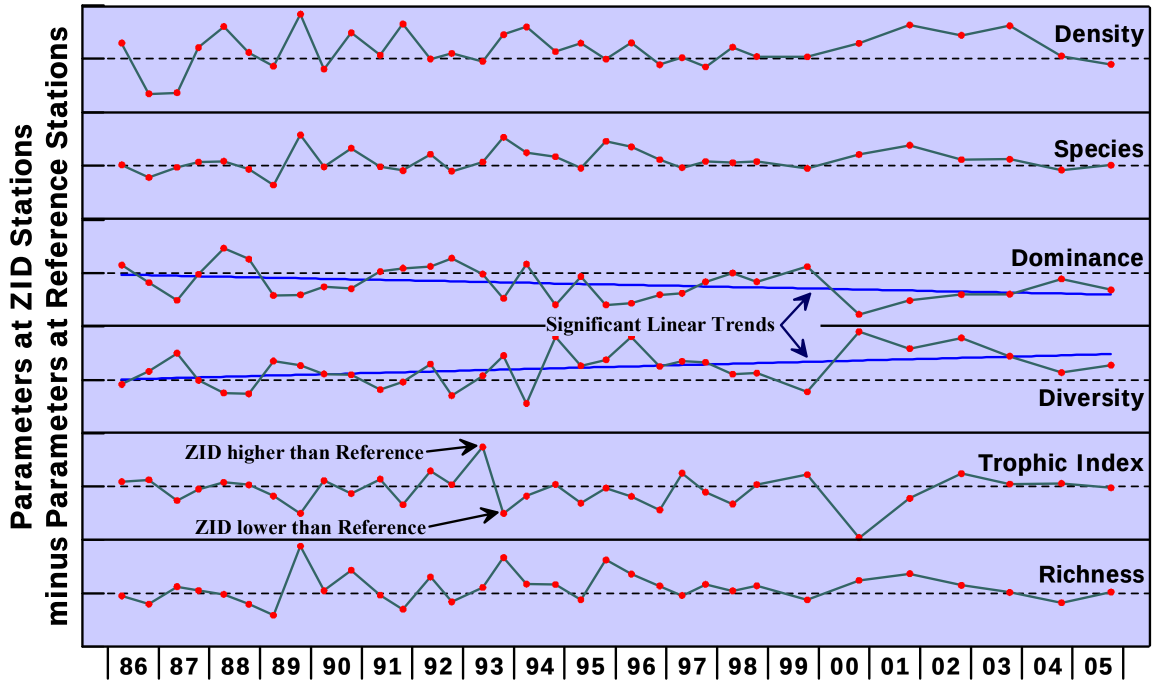

Figure 4.14 Difference in Average Infaunal Parameters between Reference and ZID Stations

Parallelism tests applied to the entire 20-year MBCSD time series discerned a statistically significant linear trend in diversity (p=0.022) and marginally in dominance (p=0.048), but not in the other indices (p>0.10). However, most of the departure from parallelism in dominance and diversity has occurred in the last decade. Prior to 1993, differences in the dominance and diversity at nearfield and distant stations did not exhibit a consistent difference (Figure 4.14). However, in all but one of the 16 surveys conducted since that time, the average diversity at stations near the outfall was higher than at the distant reference stations. As discussed in previous sections, this trend is unrelated to a contaminant buildup from effluent discharge because it is the opposite of that expected from deleterious impacts arising from exposure to contaminant loads emanating from the outfall. Similarly, the trend toward reduced dominance near the outfall is indicative of a more balanced indigenous population near the outfall. Both trends reflect a healthier infaunal community at stations close to the outfall compared to distant reference stations. Positive values, shown above the dashed zero-difference line in Figure 4.14, indicate that indices at the ZID stations were higher than at the distant reference stations. Thus, the recent trends that are evident in Figure 4.14 reflect an infaunal population near the diffuser that has become increasing more populous, diverse, and richer than at the distant reference stations, where populations have become increasingly dominated by only a few organisms.

It is also possible that the measured trends reflect a beneficial impact from the effluent discharge. The improved health of the benthic community near the outfall may result from an increased loading of suspended organic particulates that are relatively devoid of chemical contaminants. It could also arise from the physical presence of the diffuser structure itself. The structure not only influences flow within the benthic boundary layer, but its sheltering effects may provide a more hospitable environment for certain infaunal taxa (Davis et al. 1982). Finally, it is clear from Figure 2.2 that the structure acts as an artificial reef and supports a diverse community of epifaunal organisms. Localized dispersion of some of these epifaunal taxa may beneficially influence recruitment of infauna at the nearby ZID stations.

4.3.4 Factors Influencing Benthic Community Composition

The spatiotemporal trends in infaunal populations that are described in the foregoing section are opposite of those expected of a benthos that is becoming increasingly impaired by effluent discharge. Because some of these biological differences result from a somewhat anomalous reference station (1), they cannot be entirely ascribed to beneficial impacts that the effluent discharge may be having on the marine ecosystem near the diffuser structure. Nevertheless, some of the spatiotemporal gradients in the population parameters were statistically significant, and further investigation into their cause is warranted. Unfortunately, the univariate diversity indices described above provide only limited insight into the composition of infaunal communities and how they vary in space and time. Application of multivariate analysis, however, can provide the needed insight into the individual taxa that are responsible for the observed differences among the various monitoring stations. As such, multivariate analysis also lends insight into naturally occurring environmental factors that may be influencing site-specific infaunal communities.

Multivariate Analysis

Investigating changes in global community properties through the use of total abundance or various diversity indices dilutes the value of information contained in the response of individual taxa. For example, a dominant taxon may show little change in response to contaminant exposure, while a less-abundant, more pollution-sensitive taxon may exhibit marked reductions or disappear altogether, allowing a more opportunistic taxon to takes its place. Under those circumstances, an infaunal index may only exhibit a weak response to the impact, even though the community composition was conspicuously changed. A variety of strategies have been developed to compare infaunal community composition by overcoming obstacles such as how to reduce the influence of rare species, and how to simultaneously analyze tens, and sometimes hundreds, of different individual taxa. Determining which species to include or eliminate can be controversial. Rare species are inherently problematic for statistical analysis, but biologists continue to debate the efficacy of excluding taxa that may be especially sensitive to impacts, even if they are few in number.

Multivariate analysis, which simultaneously examines changes in a large number of variables, provides a far superior measure of community composition. It distills pertinent information about the community into a limited number of parameters by reducing redundant information introduced by species whose responses are highly correlated. Although multivariate analysis encompasses a broad range of techniques, principal component analysis (PCA) is used here to identify those taxa that materially participate in the observed differences in community composition during 2005. PCA arranges samples along orthogonal axes (principal components) based on species composition. The distribution of samples along each principal axis often approximates some pattern of response of the infaunal community to underlying environmental gradients. Thus, PCA condenses the complex species-abundance database into a few factors responsible for the observed variability within the infaunal community. The PCA applied here is capable of balancing the influence of rare versus dominant taxa through selection of appropriate subsample sizes (Coats et al. 1999, Trueblood et al. 1994, Grassle and Smith 1976).

Differences in Community Composition

Figure 4.15 provides a visual interpretation of the PCA that was performed on the October 2005 infaunal data. It is called a biplot because it simultaneously displays differences among stations (by the location of discrete points) and the species that differentiate those samples (shown as arrows or vectors). Thus, the biplot arrows help identify the species most responsible for the differences in community composition among the nine stations.

The distribution of samples along the first principal component (x-axis) of the biplot unambiguously distinguishes the unique infaunal communities in samples collected at Stations 1 and 8 from the infaunal community at Station 9. This axis accounted for a large portion (34%) of the dissimilarity among the nine stations and clearly showed that the community compositions at Stations 1 and 8 were distinctly isolated from that of Station 9 and the remaining stations. Specifically, samples collected at Stations 1 and 8 had a large positive offset with respective Axis-1 sample scores of 0.34 and 0.26, as indicated by the location of the red diamond (♦) and green diamond (◇). Conversely, samples collected at Station 9 had negative offsets along the first principal axis with an Axis-1 sample score of -0.23, as indicated by the location of the blue diamond (♦).

The difference in the infaunal community at Station 1 was probably due to its physical separation from the majority of the other stations. It lies 1 km north of the diffuser structure and was separated from Station 2 by 889 m. The average separation of the other benthic monitoring stations was only 100 m. Biogeographic differences within Estero Bay probably account for most, if not all, of the obvious differences in the infaunal community at this distant reference station. Nevertheless, Station 1 supported an infaunal community that was surprisingly similar to that of Station 8. This is evident from the large positive offset of the Station-8 component scores along the X axis.

Station 8 is the only station that lies offshore of the diffuser structure, and shares physical similarities to Station 1 in its grain-size composition. Because wave-induced orbital motions decrease with increasing depth, deeper stations tend to have a greater proportion of fine particulates. Figure 4.6 on Page 4-26 shows that both Station 1 and Station 8 had higher mud proportions than any of the other stations. Many benthic studies have established strong correlation between fine-sediment fractions and associated infau-

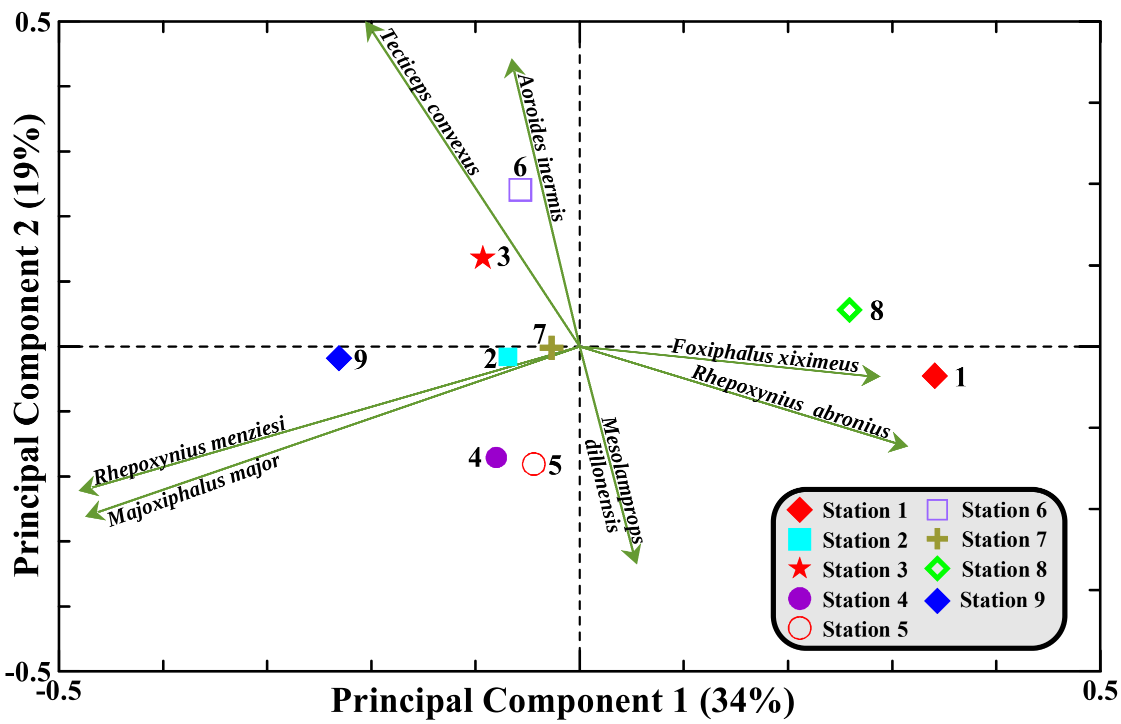

Figure 4.15 Principal Components of Infaunal Samples Collected during 2005

nal communities (Snelgrove and Butman 1994). Because Station 1 and Station 8 have elevated mud fractions, it is not surprising that these stations also support similar infaunal communities that depart from those of the other stations. The increased mud fraction at Station 8 arises, in particular, because of its greater water depth compared to the other stations. The increased mud fraction at Station 1 may arise because its location may be afforded more shelter by Point Estero from northwesterly swell (see Figure 2.1 on Page 2-2).

Conversely, Station 9 is the only station that lies shoreward of the diffuser structure. Stations 8 and 9 were added in 1999 to provide measurements at locations that were perpendicular to the north-south alignment of the other stations. Because of its location, Station 9 is likely to be exposed to greater wave-induced currents. Consequently, it has a lower mud fraction than most other stations (compare the white bar with the level of the other bars in the lower left frame Figure 4.6 on Page 4-26).

Influential Taxa

The taxa that account for the community differences at Stations 1, 8, and 9 are also evident in Figure 4.15. Biplots are designed so that the arrows point in the direction of samples with comparatively increased abundances of a given species. Thus, the arrows that point toward the right identify species that were conspicuously more abundant at Stations 1 and 8. Conversely, the arrows that point toward the left identify taxa that were comparatively rare in those samples, but were more prevalent in samples collected at Station 9. Overall, the PCA identified seven taxa that were influential in distinguishing community composition among the samples collected in October 2005. Four of these taxa were largely responsible for the separation of samples along Principal Axis 1, namely, in distinguishing the infaunal communities in samples collected at Stations 1 and 8 from the community in samples collected at Station 9. The remaining three taxa were largely responsible for the separation of samples along Principal Axis 2, namely, in distinguishing the infaunal communities at Nearfield Stations 6 and 3 from the communities in samples collected at ZID Stations 4 and 5.

The four taxa that distinguish Stations 1 and 8 from Station 9 were all phoxocephalid amphipods: *Majoxiphalus major*, *Foxiphalus xiximeus*, *Rhepoxynius menziesi*, and the closely related *Rhepoxynius abronius*. Amphipods, such as the one pictured in Figure 4.16, are small shrimp-like scavengers that eat small organic debris. These organisms are laterally composed with small eyes, small heads, and strongly arched backs. They are benthic filter feeders that are generally found burrowing within the top few centimeters of sediment. As such, they are sensitive to contamination, high organic content, and exposure to fine grained sediments. Because of their narrow habitat specificities, phoxocephalid amphipods are found only at selected depth intervals and locations along the central and southern California coast.

Figure 4.16 Photograph of an Amphipod

Figure 4.15 shows that samples collected at Stations 1 and 8 had relatively few specimens of *M. major* and *R. menziesi*, but supported relatively high abundances of both *F. xiximeus* and *R. abronius*. *R. abronius* is particularly sensitive to adverse environmental conditions, which has led to its use in bioassay testing (Ott 1986, ASTM 1992). The increased mud fraction found at Stations 1 and 8 reflects a more quiescent wave environment that is apparently better suited to *R. abronius*. However, the increase in *F. xiximeus* populations at Stations 1 and 8 was inconsistent with 2004 samples so other factors may be influencing its distribution.

In contrast, Station 9 is located closer to shore where it is exposed to more energetic bottom disturbances from shoaling waves. Correspondingly, relatively few *R. abronius* and *F. xiximeus* were found in samples collected at that station. In the absence of competition for the same ecological niche, two other closely related phoxocephalid amphipods, *R. menziesi* and *M. major*, were comparatively abundant. These groupings are generally consistent with the results of the PCA conducted in the past three years, further supporting the idea that physical characteristics, such as grain size distribution, are responsible for the differences in community structure between these stations.

Although the differences are much smaller, the possible influence of the outfall can be seen in the sample separations along the second principal axis. Namely, it appears to identify subtle differences between the infaunal community close to the outfall at ZID Stations 4 and 5, and the community at nearfield Stations 3 and 6. However, this axis only accounts for 19% of the overall variability in the infauna collected during 2005, and similar patterns were not observed in multivariate analyses conducted in previous years.

Specifically, Figure 4.15 suggests that the nearfield stations (3 and 6) had an increased abundance of an isopod (*Tecticeps convexus*) and a gammarid amphipod (*Aoroides inermis*) relative to the abundance of these taxa at ZID Stations (4 and 5). However, examination of Table E.1 in Appendix E shows that separations seen on Axis-2 are not a result of an absence of these species at the ZID Stations, so much as an increase in the populations at Stations 3 and 6. In fact, all but one of the 39 specimens of *A. inermis* were found in a single replicate sample collected at Station 6 (Table E.7). Clearly, the distribution of this species is extremely patchy, and therefore, undersampled with regard to discerning spatial differences. This hypothesis is further supported by data from 2003 and 2004. Only one specimen of *A. inermis* was encountered in the 90 benthic grab samples collected in those surveys. In any regard, the fact that the populations of *T. convexus* and *A. inermis* were not significantly altered at the ZID stations, which lie closest to the outfall, demonstrates that the discharge was not responsible for any of the perceived spatial differences in the populations of these species during 2005.

Finally, Figure 4.15 also shows that there was a slight increase in the abundance of a cumacean (*Mesolamprops dillonensis*) at the ZID stations relative to other stations. However, if this increase is associated with the outfall, then it is probably a consequence of the physical presence of the diffuser structure rather than exposure to the discharge. This is because cumacea are pollution-sensitive taxa whose populations normally decline when they are exposed to high organic loading associated with ocean outfalls (Agg et al. 1978, Roper et al. 1989, and Roper 1990). Cumacea, such as the one shown in Figure 4.17, are small crustacea with an inflated carapace, which may be ornamented with spikes or horns, a narrow abdomen, and a thin forked tail. Cumaceans burrow into the surface of clean sandy and muddy marine sediments where they obtain food by filtering water just above the sediment surface and by grazing on the surface of sand grains. The *ITI* classifies them in the Group-II trophic guild with other surface-detritus feeders. As such, their increased presence near the outfall is indicative of comparatively clean sediment conditions rather than an impaired environment.

Figure 4.17 Photograph of a Cumacean

CHAPTER 5

Conclusions and Recommendations

5.0 CONCLUSIONS AND RECOMMENDATIONS

The MBCSD Monitoring and Reporting Program was designed to evaluate the performance of the wastewater treatment plant and monitor the quality of the effluent discharged to the ocean. The offshore component of the monitoring program was intended to quantify any potential effects of the discharge on the local ecology and on the beneficial uses of the receiving waters. The reporting program, of which this report is a part, documents compliance with the discharge and receiving-water limitations specified in the NPDES discharge permit. This annual report evaluated permit compliance and documented potential discharge impacts through quantitative analyses of an extensive data set encompassing effluent constituents, receiving water quality, benthic sediment chemistry, and marine biota. Of the thousands of measurements collected as part of the monitoring program during 2005, none exceeded the discharge limitations.

5.1 TREATMENT PLANT

The Morro Bay/Cayucos Wastewater Treatment Plant has been operating under the provisions of a 301(h)-modified NPDES permit since 1986, with 2005 marking the twentieth year of consistently high performance by this treatment facility. There is no evidence of a decline in treatment performance in the two-decades of annual averages shown in Table 5.1. In fact, many of the best measures of treatment performance were achieved in the most-recent two years of operation. For example, the average effluent TSS concentration in 2005 (24.3 mg/L) was the second lowest on record, and was far lower than any other average concentration except for the 21.8-mg/L concentration measured in 1987, which was an El Niño year. The next lowest effluent TSS concentration of 28.9 mg/L was measured in 2004. Similarly, the highest average solids removal rate (93.3%) ever recorded in two decades of operation was achieved in 2005, with 2004 removal rates a close third at 91.3% just behind the 1987 removal rates. Combined with the low total flow in 2005, the plant's solids-removal process resulted in the lowest annual mass emission of suspended solids (42 MT) recorded since monitoring began in 1986. This is only about one-fifth of the solids discharge permitted under the NPDES permit. 2004 was a close second with only 44 MT of solids discharged over the entire year.

O&G and BOD concentrations and emissions were also low in 2004 and 2005, and removal rates were high compared to average rates achieved over the two decades of monitoring. Averages computed over the two decades of monitoring are shown in bold at the bottom of Table 5.1, just above the row that lists the permit limitations. Across the board, effluent TSS, O&G and BOD concentrations and emissions were lower in 2005 than the monitoring-program averages despite higher-than-average influent concentrations for these same constituents. Accordingly, 2005 removal rates for these three constituents were among the highest measured in any year since monitoring began, and were markedly higher than the average removal rate for the monitoring program overall. Mass emissions were particularly low in 2005 due to the lower-than-average flow rate in 2005. Finally, average effluent turbidity at 23 NTU was the lowest on record, tying with the average turbidity measured in 1987.

5.1.1 Overall Performance

Irrespective of comparisons with historical performance, the wastewater parameters reported in 2005 demonstrate that the treatment process was highly effective. As a direct result of its effectiveness, effluent quality was well above the established permit criteria. The average suspended-solids removal rate was 18% above the permitted minimum, and the BOD removal rate was more than two and a half times the minimum permitted rate of 30%. The treatment process was so efficient that the resulting effluent TSS were well below the minimum 60 mg/L during all 12 months in 2005, and BOD concentrations were well below this minimum in all but three months. This is an effluent concentration so low that the required removal rates do not apply. Moreover, the plant routinely achieved treatment levels that exceeded the requirements for full secondary treatment. For example, monthly solids-removal rates surpassed the 85% full-secondary criterion during all of 2005. Other effluent constituents were also low during 2005. The discharge of settleable solids was imperceptible, and monthly averages of effluent turbidity, pH, ammonia, and coliform density remained far below the applicable permit limitations.

The benefits of past efforts to reduce rainwater inflow and groundwater infiltration (I&I), and to more accurately measure plant flow, are evident in the record of annual average flow rates listed in Table 5.1. Reported flow during 2005 was the fourth lowest on record, following the record low flows in the three prior years. In 2005, 73 million fewer gallons were reported to have flowed through the plant than the average annual throughput. Some of the flow reduction in 2005 was an artifact of improvements in the accuracy of flow measurements.

Table 5.1 Average Annual Wastewater Parameters

		Suspended Solids				Biochemical Oxygen Demand			
Year	Flow (MGD)	Influent (mg/L)	Effluent (mg/L)	Removal (percent)	Emission (MT)	Influent (mg/L)	Effluent (mg/L)	Removal (percent)	Emission (MT)
1986	1.42	332	32.8	89.8	64	235	77.0	67.2	151
1987	1.51	274	21.8	92.0	45	257	52.0	79.8	108
1988	1.51	397	29.8	90.0	62	242	43.9	81.9	92
1989	1.46	321	37.3	88.4	75	259	69.8	73.1	141
1990	1.38	345	36.0	89.6	69	261	75.7	71.0	144
1991	1.28	280	30.5	89.1	54	236	66.9	71.6	118
1992	1.41	310	43.0	86.3	84	224	59.3	73.5	116
1993	1.54	339	33.0	89.6	70	222	39.0	81.9	83
1994	1.38	310	32.0	89.4	61	249	33.0	86.4	63
1995	1.55	270	30.6	87.6	69	208	31.4	83.9	67
1996	1.55	344	33.1	89.9	70	241	35.7	85.0	73
1997	1.64	283	36.0	86.6	79	231	38.6	83.0	85
1998	1.95	236	38.8	83.9	101	216	39.1	81.5	99
1999	1.68	386	44.0	86.7	102	287	49.5	82.5	118
2000	1.77	337	37.4	87.5	91	271	50.3	81.1	125
2001	1.48	450	37.6	89.5	74	396	62.7	83.1	127
2002	1.14	374	49.2	86.0	77	386	67.5	82.4	101
2003	1.06	314	39.2	86.7	56	311	56.3	81.3	83
2004	1.09	354	28.9	91.3	44	336	53.3	83.8	81
2005	1.25	373	24.3	93.3	42	303	49.8	83.0	88
Mean	**1.45**	**331**	**34.8**	**88.7**	**69**	**269**	**52.5**	**79.9**	**103**
Limitation	**2.06**		**70.0**	**75.0**	**199**		**120.0**	**30.0**	**342**

Table 5.1 Average Annual Wastewater Parameters (continued)

Year	Oil and Grease Influent (mg/L)	Oil and Grease Effluent (mg/L)	Oil and Grease Removal (percent)	Oil and Grease Mass Emission (MT)	Turbidity (NTU)	pH	Acute Toxicity (TUa)	Chronic Toxicity (TUc)	Ammonia as NH_3-N (mg/L)
1986	64	13.8	78.4	27	26	7.7	1.3[1]		18
1987	44	6.2	85.9	13	23	7.5	1.9[1]		
1988	38	6.3	83.4	13	40	7.5	1.2[1]		
1989	28	6.1	78.2	12	49	7.4	1.1[1]		26
1990	34	8.5	75.0	16	55	7.4	1.4[1]		26
1991	73	6.9	90.5	12	50	7.3	1.4[1]		18
1992	33	5.3	83.9	10	56	7.3	1.2[1]		9
1993	26	6.0	76.9	13	43	7.4	1.7[1]	19.42[2]	20
1994	60	4.1	93.2	8	36	7.5	1.3[1]	4.37	27
1995	63	5.1	91.9	11	32	7.5	0.7	4.35	23
1996	52	7.9	84.8	17	34	7.7	0.5	4.83	23
1997	49	5.3	89.2	12	32	7.7	0.4	7.80	23
1998	51	5.4	89.4	15	34	7.6	0.1	7.80	19
1999	52	6.2	88.1	14	48	7.5	0.0	5.00	25
2000	74	5.5	92.6	13	39	7.5	0.0	5.60	24
2001	47	4.6	90.2	9	41	7.4	0.1	5.60	28
2002	39	4.4	88.7	7	41	7.5	0.0	4.98	31
2003	44	5.3	87.9	7	34	7.5	0.9[1]	7.80	27
2004	47	3.7	92.0	6	26	7.5	1.4[1]	5.60	29
2005	62	4.4	92.9	8	23	7.6	1.2[1]	5.60	27
Mean	**49**	**6.1**	**86.7**	**12**	**38**	**7.5**	**0.9**	**6.83**	**24**
Limitation		**25.0**			**75**	**6-9**	**1.5[3]**	**134.00**	**322**

[1] Interference from ammonia artificially increased the reported acute toxicity.
[2] Screening bioassay of three marine species
[3] The acute-toxicity limitation that is consistent with the current California Ocean Plan is 4.3.

Improved accuracy notwithstanding, a portion of the flow reduction was genuine, and due to reductions in the fraction of flow arising from I&I. An increased effort to reduce I&I in recent years has resulted in a decrease in the amount of relatively clean rain and groundwater that is unnecessarily treated at the WWTP. These I&I reductions have reduced the total volume of flow processed by the plant, thereby increasing the proportion of wastewater of sewage origin that is treated by the plant.

5.1.2 Isolated Events

Along with the high overall performance of the treatment process during 2005, there were eleven isolated instances of noticeable short-term changes in wastewater parameters. These anomalous measurements were all related to specific external events or component failures that occurred during 2005. The fact that these abnormal wastewater measurements were few in number and did not occur randomly demonstrates the overall effectiveness of the treatment process, and the high degree of control that is exercised by the plant personnel.

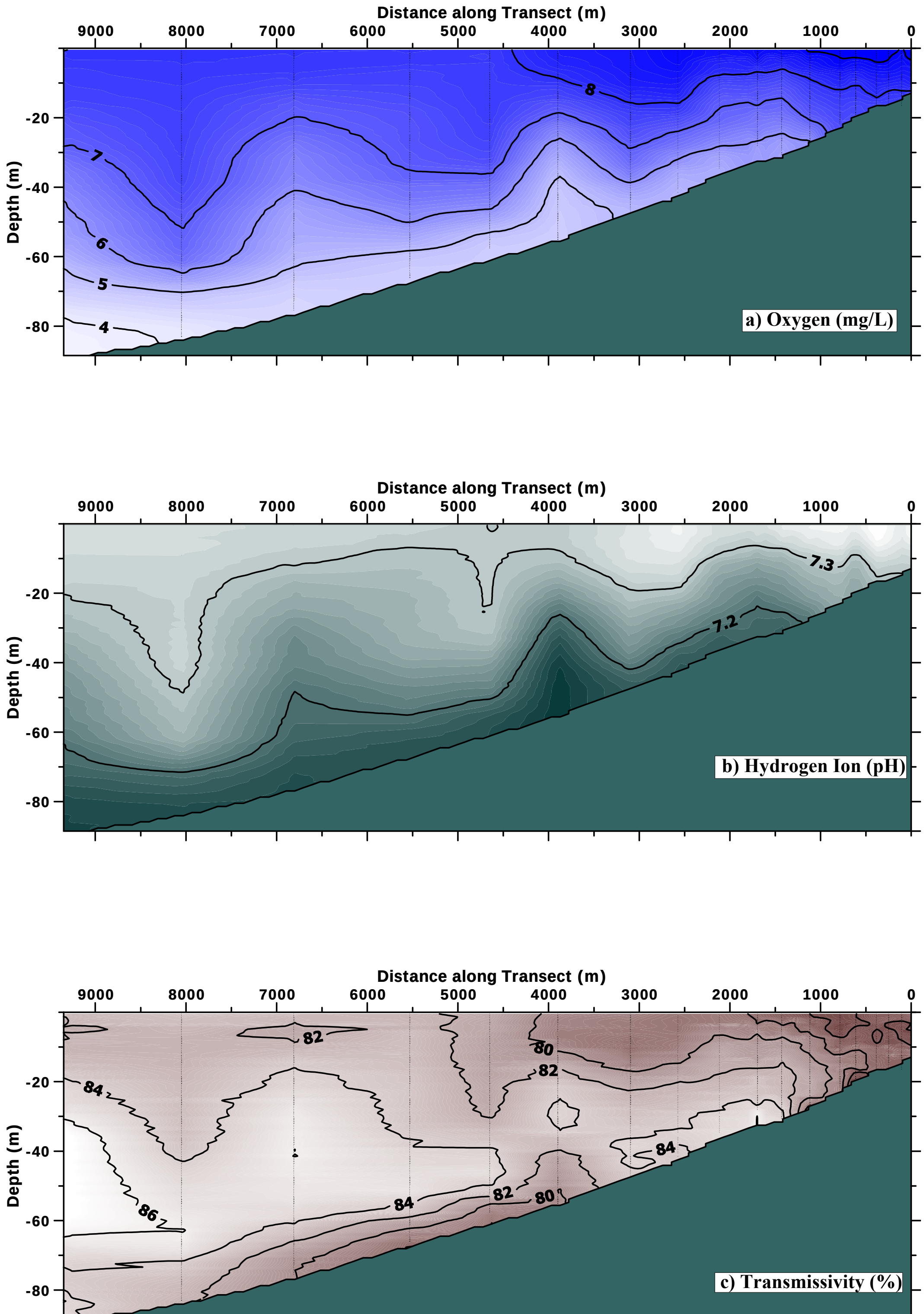

Figure 3.6 Cross-Shore Vertical Sections of: a) Oxygen, b) pH, and c) Transmissivity Collected across Estero Bay on 15 April 1999

All but one of these events resulted from short-term increases in flow resulting from rainwater inflow or population increases in the service area during holidays. The remaining event occurred when improvements to the disinfection system piping required manual infusion of chlorine, which resulted in chlorine concentrations that were slightly higher than the norm. In addition to implementing an vigilent and proactive program of preventative maintenance, plant personnel acted quickly to correct unforeseeable mechanical failures of treatment-plant components that occasionally occurred throughout 2005.

In most of the years since monitoring began in 1986, three major influences identifiably increased measured plant flow. These were overtotalization, I&I, and short-term population increases in the service area. Historically, the most influential factor was the artificially increased flow that resulted from overtotalization. Overtotalization was particularly significant prior to 2002, when flow rates were reported using only the effluent flow meter. This resulted in flow rates that were overestimated by as much as 25% and average flow over the two-decade record should be closer to 1.2 MGD rather than 1.45 MGD. Because the flow rates were overestimated prior to 2002, the annual mass emissions were also overestimated. Consequently, the mean emission, shown in bold at the bottom of Table 5.1, is slightly inflated. Specifically, the reported average annual TSS emission of 71 MT was actually closer to 58 MT. Similarly, the average annual BOD emission that is reported to be 104 MT was closer to 86 MT, and the reported 12 MT of O&G discharged on average was closer to 10 MT.

In 2002, flow overtotalization was substantially reduced with the use of the much more accurate influent flow meter. However, flow was still occasionally overtotalized due to aberrant influent flow readings that occur when the influent flume becomes temporarily surcharged after water backs up from the wetwell at the plant headworks. In 2004, judicious use of corrected effluent flow totals on those rare occasions effectively eliminated flow overtotalization. During 2005, there was only one instance when the corrected effluent flow totals were used in place of influent readings.

The second type of event that identifiably increased flow rates arose from rainwater inflow and groundwater infiltration. I&I was responsible for at least 18.6% of the flow measured during 2005. Brief but marked increases in flow rate were directly correlated with rainstorms. Approximately one million gallons of rainwater flowed into the collection system for every inch of rain that fell during 2005. This inflow rate was approximately double the rate determined in years prior to 2004. The increase in the inflow rate may be related to damage somewhere in the collection system that may have occurred during a major earthquake at the end of 2003.

Over longer periods, infiltration of groundwater into the collection system observably increased plant throughput during 2005. Because inflow from rainstorm events was brief in nature, it contributed much less than the volume generated by groundwater infiltration. Also, at 0.205 MGD, the lower-bound estimate for the average infiltration reported for 2005 may be unrealistically low. It was computed from the amplitude of low-frequency flow variations that extend above a baseline flow level. In the absence of other information, this baseline flow level was simply selected as the lowest flow of the year. However, in 2005, the baseline flow was 0.154 MGD higher than in 2002, which suggests that infiltration was at least 0.359 MGD, or 28% of the flow in 2005. Moreover, even the lower baseline in 2002 probably did not constitute pure wastewater of sewage origin. Instead, it may have also included substantial contributions from groundwater infiltration.

In any regard, the presence of significant I&I contributions gives the mistaken impression that the treatment plant was operating at approximately 60% of its dry-weather design capacity in 2005. I&I contributions are relatively devoid of organic solids, and consequently, do not tax the treatment process to the same extent as pure sewage. In reality, after accounting for I&I, wastewater of purely sewage origin constituted less than 55% of the plant's design capacity during the dry season.

The third type of identifiable event that affects flow arises from large population fluctuations within the service area. Short-term population increases on weekends and holidays accounted for approximately 8.2% of the total flow through the treatment plant during 2005. In comparison, longer-term population increases during the summer tourist season accounted for approximately 1.8% of the annual flow. Clearly, the predominantly tourist-based economies of Cayucos and the City of Morro Bay are responsible for a large portion of the population-related fluctuations in flow rates. Baseline flow through the treatment plant, without the additional flow contributed by I&I, and tourism, represented less than 72% of the total flow measured during 2005.

5.1.3 Effluent Contaminants

The treatment process is primarily designed to disinfect the effluent and to remove organic particulates from the wastewater stream. It is not designed to eliminate chemical contaminants that are dissolved in wastewater. Instead, the vigorous pollution prevention program of the MBCSD limits the introduction of chemical contaminants at the source, before they enter the collection system. The multifaceted pollution prevention program includes public education, an onsite hazardous waste collection facility, source identification, and inspections of industrial users. Ninety percent of the sewage is generated by domestic users, and the remaining contribution from businesses comes from non-industrial users or light industry, which generate wastewater similar to that of domestic sources but on a larger scale. In the absence of heavy industry within the service area, there is a concomitant lack of industrial pollutants within the MBCSD wastewater.

As in prior years, a few common metals and ubiquitous chemical compounds were found in low concentrations within effluent and biosolid samples collected during 2005. Of the 167 chemical compounds tested in effluent, only eleven had quantifiable concentrations. The measured concentrations for each of these detected compounds were well below applicable NPDES discharge limits. In most cases, the concentrations were orders of magnitude lower. The associated mass emissions were also well below the goals identified in the NPDES Monitoring and Reporting Program for MBCSD.

Detected compounds included five trace metals (arsenic, copper, lead, selenium, and zinc), cyanide, a common phthalate compound (bis 2-ethylhexyl phthalate), and dioxin. The ever present ammonia, chlorine, and radionuclide constituents represented the three remaining compounds that were quantified in the 2005 effluent samples. Because of its origin, ammonia is a regularly occurring constituent in sewage-derived wastewater. However, peak ammonia concentrations were 24 times lower than the permitted instantaneous limit imposed for the protection of marine aquatic life. In contrast to ammonia, chlorine is introduced during the disinfection process at the treatment plant. If it is not fully buffered by the introduction of sodium bisulfite, then a residual amount of chlorine can occasionally appear in effluent samples. Careful control of the disinfection process throughout 2005 maintained residual chlorine at undetectable levels in 95% of the daily samples. None of the remaining samples had chlorine concentrations that exceeded the permitted limits. As with other common effluent constituents, the presence of radionuclides, as measured by radioactivity, was also found to be low. During 2005, radioactivity was well within limits established for drinking water standards and was similar to historical levels.

Trace metals occur naturally within the mineralogy of sediments along the central California coast, although arsenic, copper, lead, and zinc also enter the collection system through internal corrosion of household plumbing systems. Regardless of their source, the concentrations of these metals were low, and well below the limitations specified in the NPDES discharge permit.

Similarly, bis 2-ethylhexyl phthalate (BEHP) , was found at low concentrations in the 2005 effluent samples, as has been the case in most effluent and biosolid samples collected in the past. This chemical would have to be at least 47-times more concentrated in effluent before it would be of a human-health concern. Cyanide was also detected in both the January and July effluent samples, but its concentrations were very low, and only slightly above its PQL. The measured concentrations were almost 6-times lower than the permit limitation and therefore, not of ecological concern. Although cyanide is not normally detected in effluent samples, it is commonly found in low concentrations in biosolids.

Dioxin was another ubiquitous chemical detected in the July-2005 effluent sample. Low concentrations of dioxin have been detected in approximately 82% of the effluent samples tested over the last decade. Dioxins are chemically stable and do not easily break down into other compounds. Therefore, once produced, dioxins persist in the environment and are found almost everywhere. Extremely small concentrations, less than a part-per-trillion, of various dioxin congeners are often found in effluent discharged from publicly owned treatment works, where they are formed during the chlorination process. The measured dioxin concentration in the July 2005 sample was below the discharge permit limit and the computed mass emission was below the discharge goal.

As with effluent samples, chemical analyses of biosolid samples detected only very low concentrations of fifteen ubiquitous trace metals and BEHP. All measured trace-metal concentrations were low, and well below regulatory limits that would make the biosolids hazardous or unsuitable for composting and land application.

Because of low overall contaminant concentrations in effluent, bioassays conducted over the past decade consistently demonstrate that the MBCSD effluent has low toxicity to marine organisms. The chronic bioassays conducted in 2005 confirmed the benign nature of MBCSD effluent. Chronic bioassays are far more accurate and sensitive than the acute bioassays, which were also conducted on effluent samples in 2005. Although all toxicity measurements in 2005 were below permit limits, the acute bioassays were conducted using new protocols that introduced significant interference from ammonia. This interference artificially elevated the acute toxicity levels reported in 2005.

5.2 RECEIVING WATERS

The receiving-water environment was monitored on a quarterly basis to evaluate the oceanographic conditions near the outfall, particularly with respect to any adverse impacts from wastewater discharge. Comparison of water quality at the boundary of the zone-of-initial dilution with gradient areas beyond the dilution zone documented compliance with the receiving-water objectives of the California Ocean Plan as specified in the NPDES discharge permit. Extremely sensitive sensors provided a detailed picture of water quality during the four surveys conducted during 2005. Precise navigation and high-resolution data on light transmittance, density, temperature, salinity, pH, and dissolved oxygen were used to assess spatial variations within receiving waters.

During all four surveys, small anomalies in water properties associated with the submerged wastewater plume were detected. In all cases, the water-quality fluctuations were either restricted to the zone of initial dilution, were generated by the upward displacement of ambient seawater and not the presence of wastewater constituents, or were insignificant compared to the larger ambient variations that result from natural oceanographic processes. None of the observed conditions suggested that unmixed wastewater was significantly impacting receiving waters beyond the zone of initial dilution. On the contrary, computed dilution factors demonstrated that the diffuser structure had dispersed the wastewater to a much greater extent than that predicted by modeling conducted during the outfall's design. As with prior monitoring, water quality parameters measured in the year-2005 surveys confirmed that the diffuser was operating efficiently, and that wastewater was achieving a high level of dilution immediately following discharge from the diffuser ports.

5.3 BOTTOM SEDIMENTS

The monitoring program has evaluated physical, chemical, and biological conditions within the benthic sediments around the outfall for almost two decades. Temporal fluctuations in these factors have been directly related to natural influences, the largest of which occur on seasonal and interannual time scales. The more notable faunal variations have involved interannual population fluxes within individual taxa. For example, large increases in the abundance of the Pacific Sand Dollar (*Dendraster excentricus*) occurred in 1989, 1991, and 1999. During these recruitment years, Sand Dollar populations dominated the infaunal community. Their marked population increases coincided with the well-recognized global climate changes known as El Niño. These major changes affected all of the benthic monitoring stations, including reference sites, and thus could be clearly ascribed to inherent oceanographic variability rather than the discharge of effluent.

To test for outfall-related effects, a large number of biological indices and parameters were computed from an enumeration of the 142,000 specimens collected over the two decades of benthic monitoring. None of these parameters exhibited statistically significant spatial distributions related to deleterious impacts from effluent discharge, or long-term spatiotemporal trends that would be indicative of an increasingly degraded benthic habitat near the outfall. Not only were the measured spatial differences small compared to inherent sampling variability, but they were also overwhelmed by seasonal and interannual changes in community structure resulting from natural environmental oscillations, such as El Niño. Despite large temporal fluctuations in the infaunal community, its health has remained uniformly high at all stations throughout the two decades of monitoring. The benthic environment surrounding the outfall has always been populated by an infaunal community dominated by pollution-sensitive suspension-feeding organisms.

Several physicochemical properties have exhibited statistically significant increases since benthic monitoring began in 1986. The increases are clearly unrelated to effluent discharge, however, because they are evident at all monitoring stations, including the distant reference stations. In addition, the annual mass emission of trace metals discharged by the treatment plant is far too small to account for the observed increases in sediment concentrations. Based on supplemental sediment analyses conducted in October 2000, erosion from abandoned onshore chromite mines represents the most likely source for the increasing metal concentrations in offshore sediments.

In the past, most sediment metal concentrations have been comparable to those determined in nearby benthic environments, such as Port San Luis and Moss Landing (NOAA 1991a), or within Morro Bay itself

(Tenera and Marine Research Specialists 1997). Accordingly, contaminant concentrations measured in bulk sediment samples collected within northern Estero Bay, both near and far from the outfall, have been below thresholds considered harmful to marine biota.

5.4 PUBLIC HEALTH

Water recreation is an integral part of the environment near the City of Morro Bay and Cayucos. The ocean waters within the surfzone along Atascadero State Beach are frequently utilized by the public. To ensure that the public health is not being adversely affected by wastewater discharge, the MBCSD regularly samples the surfzone for coliform bacteria. Over the years, this surfzone monitoring has detected many instances of elevated levels of indigenous and enteric bacteria. However, these instances of elevated bacteria were unrelated to effluent discharged from the treatment plant. The plant disinfects all wastewater before it is discharged. Consequently, instances of elevated coliform densities measured in surfzone samples were much higher than the total coliform levels measured concurrently in effluent samples at the treatment plant. Moreover, effluent is discharged through a diffuser structure that rapidly mixes with seawater immediately upon discharge. This results in seawater concentrations that are less than 0.75% of the original contaminant concentrations measured within effluent. This occurs within 15 m of the discharge, which lies 0.5 miles from shore. There is little likelihood that effluent impinges on the adjacent shoreline half a mile away, especially considering that the prevailing flow is along the shoreline rather than towards the adjacent shoreline. Finally, the disinfected effluent is devoid of other bacterial pathogens. Tissues from mussels that were collected close to the outfall in 2003 and 2004 have been tested for a broad suite of bacterial pathogens and none were detected.

Instead, decades of monitoring have clearly demonstrated that non-point sources of microbial contamination have been responsible for the occasional measurements of elevated surfzone bacteria. Terrestrial, non-point-source bacterial contamination from runoff is a well-recognized problem along the coast of central and southern California because infrequent rains allow contaminants to build up during long dry spells. Accordingly, consistently high levels (>1600 MPN/100-ml) of total and fecal coliform organisms have been detected at the mouth of Morro Creek during periods of river outflow. Other areas of the beach seldom show elevated coliform levels, except during periods of rainfall or high surf. Non-point bacterial sources on the beach cause surfzone coliform levels to elevate when high surf entrains fecal material from seabirds and other animals that has accumulated in the supratidal zone following an extended period of low tide and surf. These well-recognized sources of surfzone coliform far outweigh any conceivable contribution from disinfected effluent that is discharged far offshore.

5.5 RECOMMENDATIONS

This annual report summarizes two decades of monitoring data. These data consistently demonstrate that the MBCSD WWTP has been operating as it was designed, and that the discharge of effluent has not adversely impacted the marine environment within Estero Bay. Although marked improvements to the monitoring program have substantially increased its capacity to detect minute environmental impacts, conclusions concerning the lack of impacts have continued to hold into 2005. Many of the monitoring improvements were based on recommendations in early annual reports that were ultimately incorporated in the current NPDES discharge permit. Also, the accuracy of flow measurements markedly improved after the new influent flow meter was commissioned in 2001. Additional improvements to flow measurements were realized in 2003, after implementation of a 2002 recommendation to use adjusted effluent flow measurements when the influent flow meter was compromised.

In contrast to prior annual reports, most of the recommendations listed below deal with potential improvements to the monitoring program rather than with improvements to plant performance. The efficacy of the treatment process was near an all-time high in 2005. Consequently, many of the recommendations listed below arise from a review of the monitoring program that was conducted in 2003 (MRS 2003a). Analysis of 2005 monitoring data further strengthens and confirms the veracity of those previous recommendations.

- **Eliminate acute toxicity bioassays.** With the RWQCB-directed change in analysis protocols in April 2003, acute bioassays no longer accurately reflect the actual toxicity of the MBCSD effluent. This is solely because of toxic ammonia deionization that occurs in test chambers and artificially increases measured toxicity. This deionization does not occur in the marine environment. The current COP no longer requires acute toxicity tests in lieu of the more accurate chronic toxicity tests that the MBCSD already conducts on its effluent.

- **Reduce the frequency of effluent chemical analyses.** Most effluent chemical concentrations measured over the last two decades have been consistently well below NPDES discharge requirements. Given the historically high level of compliance, sampling on an annual basis or at longer intervals will sufficiently demonstrate continued compliance. This conclusion was demonstrated by a quantitative statistical assessment of historical contaminant measurements based on a Reasonable Potential Analysis using the methodology that is currently recommended by the SWRCB (MRS 2004b, MBCSD 2004).

- **Revise the benthic sampling program.** Station 1 should be eliminated from the suite of benthic monitoring stations. Its current location is too far north of the other monitoring stations to provide a representative control. Also, sampling at cross-shore stations (8 and 9) should be eliminated. Variation due to the depth differences at these stations unnecessarily confounds impact assessments. Finally, chemical analyses for synthetic organic compounds should be eliminated or reduced to once in the life of the permit. After nearly two decades of monitoring, these compounds have not been detected in benthic sediments, and are rarely if ever detected in effluent samples. When they are detected, they are at extremely low concentrations. Unlike trace metals, these volatile compounds are not likely to preferentially accumulate in benthic sediments. In lieu of eliminating sampling at Stations 1, 8, and 9, triplicate chemistry samples should be collected composited at each of the remaining stations, rather than the single sample that is currently collected. This will enhance the stability and reliability of the measured chemical concentrations.

- **Eliminate the requirement for sulfide analysis of benthic samples.** High-resolution sulfide analyses have been performed on porewater samples on three separate sampling occasions, in 2003, 2004, and 2005. None of the 27 samples contained detectable sulfide concentrations. Elevated sulfide concentrations in porewater are usually restricted to sheltered harbor and estuarine environments, where there is a high fraction of organic constituents within the sediments. Often these benthic environments are also hypoxic. This is not the case for the coarse sand sediments surrounding the MBCSD outfall, which are intensively reworked by waves and currents in the open ocean.

- **Revise the water-column sampling program.** Collecting vertical CTD casts at discrete stations is time consuming and does not provide sufficient lateral resolution to delineate the extremely limited extent of the effluent plume. The traditional vertical sections generated by adjacent CTD casts are of comparatively little use because they do not accurately characterize the effluent discharge, and can provide misleading impressions of the lateral extent of the discharge footprint. Even the closely spaced stations used in the current monitoring design are located too far apart to adequately delineate the highly localized effluent plume. In the current design, the distance between adjacent stations is at the limit of the accuracy of available DGPS navigation. In contrast, the advantages of continuous horizontal sampling using a towed CTD have been demonstrated in past surveys. Revisions to the up-coming NPDES monitoring and reporting program for the MBCSD permit should eliminate all vertical CTD casts except those associated with a single along-shore transect. Wastewater is preferentially transported in an along-shore direction, and a single along-shore transect will adequately define the ambient vertical structure. A towed-CTD survey should be an added requirement. The towed survey should be conducted at two depth levels. One survey should be conducted immediately above the benthic boundary layer, approximately 5 m above the bottom. The other towed survey should be conducted in the upper water column, near the base of the shallow thermocline. Each should include at least five passes over the diffuser structure. Vessel speed and sampling rates should be sufficient to collect at least one sample for every meter traversed.

- **Specify water-quality limits applicable to wastewater constituents.** Chapter 3 of this report demonstrates that most discharge-induced changes in seawater properties are the result of the upward displacement of ambient bottom water. Generally, these discharge-related changes are not the result of the presence of dilute effluent constituents. Waste discharge requirements that are to be specified in the upcoming revisions to the NPDES discharge permit should distinguish between these two processes, as does the COP. The same constraints and limitations that apply to wastewater contaminants should not apply to ambient seawater that is simply moved to a different location within the water column.

- **Conduct surfzone monitoring only when effluent coliform is elevated.** Surfzone samples are heavily confounded by bacterial contamination from non-point sources, and lend no insight into discharge-related impacts to seawater quality, the efficacy of the treatment process, or the operational performance of the diffuser structure. Because effluent is disinfected, coliform densities are rarely detected within effluent samples themselves. Even if viable bacteria were not totally eliminated by disinfection, dispersion within the 15-m ZID would be enough to reduce concentrations below the detection limit. This has been confirmed by the absence of bacterial pathogens within mussel tissue collected within the ZID during 2003 and 2004. Even without the confounding influence of non-point source contamination, surfzone stations are located too far from the discharge to provide any insight into discharge-induced bacterial levels within receiving waters.

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CHAPTER 6

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6.0 LITERATURE CITED

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APPENDIX C

Effluent and Biosolids Chemistry

APPENDIX C

Analytical Results for Effluent and Biosolid Samples Collected during 2005

Table C.1 Effluent Monitoring Results

Constituent	Units	Method	Concentration			Mass Emission	
			Permit Limit[a]	January-June[b]	July-December[c]	Loading (kg)	Goal (kg/yr)
Miscellaneous Parameters							
Flow (quarterly)	Mega-L/Qtr	NA[d]	712	**502.9** **424.3**	**433.5** **371.8**	NA	NA
Acute Toxicity (quarterly): *Pimephales promelas*	TUa	EPA-821-R-02-012[e]	1.5	**0.76** **1.41**	**1.41** **1.41**	NA	NA
Methyl tert-butyl ether (MTBE)	µg/L (ppb)	624	NA	<1[f]	<0.5	<0.8	NA
Asbestos (>10 µm & Total)	MFL	100.2	NA	—[g]	<2	NA	NA
Acenaphthene	µg/L (ppb)	625	NA	<2	<2	<3.47	NA
Aniline	µg/L (ppb)	625	NA	<5	<5	<8.66	NA
Benzoic Acid	µg/L (ppb)	625	NA	<10	<10	<17.3	NA
Benzyl Alcohol	µg/L (ppb)	625	NA	<2	<2	<3.47	NA
Butylbenzyl Phthalate	µg/L (ppb)	625	NA	<2	<2	<3.47	NA
4-Bromophenyl phenyl ether	µg/L (ppb)	625	NA	<2	<2	<3.47	NA
4-Chloroaniline	µg/L (ppb)	625	NA	<2	<2	<3.47	NA
2-Chloronaphthalene	µg/L (ppb)	625	NA	<2	<2	<3.47	NA
4-Chlorophenyl phenyl ether	µg/L (ppb)	625	NA	<2	<2	<3.47	NA
2,6-Dinitrotoluene	µg/L (ppb)	625	NA	<2	<2	<3.47	NA
Di-n-octylphthalate	µg/L (ppb)	625	NA	<2	<2	<3.47	NA
Endrin Aldehyde	µg/L (ppb)	608	NA	<0.005	<0.01	<0.0127	NA
2-Methylnaphthalene	µg/L (ppb)	625	NA	<2	<2	<3.47	NA
Naphthalene	µg/L (ppb)	625	NA	<2	<2	<3.47	NA
2-Naphthylamine	µg/L (ppb)	625	NA	<20	<20	<34.7	NA
2-Nitroaniline	µg/L (ppb)	625	NA	<2	<2	<3.47	NA
3-Nitroaniline	µg/L (ppb)	625	NA	<2	<2	<3.47	NA
4-Nitroaniline	µg/L (ppb)	625	NA	<5	<5	<8.66	NA
N-Nitrosodi-n-propylamine	µg/L (ppb)	625	NA	<2	<2	<3.47	NA
1,2,4-Trichlorobenzene	µg/L (ppb)	625	NA	<2	<2	<3.47	NA
4-Chloro-3-methylphenol	µg/L (ppb)	625	NA	<5	<5	<8.66	NA
2-Chlorophenol	µg/L (ppb)	625	NA	<2	<2	<3.47	NA
2,4-Dichlorophenol	µg/L (ppb)	625	NA	<2	<2	<3.47	NA
2,4-Dimethylphenol	µg/L (ppb)	625	NA	<2	<2	<3.47	NA
2-Methylphenol	µg/L (ppb)	625	NA	<2	<2	<3.47	NA
4-Methylphenol	µg/L (ppb)	625	NA	<2	**≈0.74**[h]	<2.45	NA

[a] Sampling statistic associated with lowest permitted limit, *viz.* either six-month median or 30-day average
[b] Chemical samples were collected on 12 January 2005, the chronic bioassay sample was collected on 10 January 2005, and four acute bioassay samples were collected on 10 January through 13 January 2005.
[c] Chemical samples were collected on 13 July 2005, the radioactivity sample was collected on 15 July 2005, the chronic bioassay sample was collected on 11 July 2005, and four acute bioassay samples were collected on 11 July through 14 July 2005.
[d] Not applicable or not specified in NPDES discharge permit
[e] Test results were artificially elevated by ammonia interference
[f] "<" indicates the substance was not detected at the indicated Practical Quantification Limit (PQL)
[g] Analysis not required
[h] "≈" indicates the substance was detected above the method detection limit but could not be accurately quantified because it was below the PQL

Table C.1 Effluent Monitoring Results

Constituent	Units	Method	Concentration			Mass Emission	
			Permit Limit[a]	January-June[b]	July-December[c]	Loading (kg)	Goal (kg/yr)
2-Nitrophenol	µg/L (ppb)	625	NA	<2	<2	<3.47	NA
4-Nitrophenol	µg/L (ppb)	625	NA	<2	<2	<3.47	NA
Pentachlorophenol	µg/L (ppb)	625	NA	<10	<10	<17.3	NA
Phenol	µg/L (ppb)	625	NA	<2	<2	<3.47	NA
2,4,5-Trichlorophenol	µg/L (ppb)	625	NA	<5	<5	<8.66	NA
Chloroethane	µg/L (ppb)	624	NA	<0.5	<0.5	<0.866	NA
1,1-Dichloroethane	µg/L (ppb)	624	NA	<0.5	<0.5	<0.866	NA
trans-1,2-Dichloroethene	µg/L (ppb)	624	NA	<0.5	<0.5	<0.866	NA
1,2-Dichloropropane	µg/L (ppb)	624	NA	<0.5	<0.5	<0.866	NA
Trichlorofluoromethane	µg/L (ppb)	624	NA	<0.5	<0.5	<0.866	NA
Total Xylenes	µg/L (ppb)	624	NA	<1	**≈0.58**	<1.39	NA
Methoxychlor	µg/L (ppb)	608	NA	<0.01	<0.005	<0.0133	NA
Azinphos Methyl	µg/L (ppb)	8140	NA	—	<0.2	<0.347	NA
Bolstar	µg/L (ppb)	8140	NA	—	<0.2	<0.347	NA
Chlorpyrifos	µg/L (ppb)	8140	NA	—	<0.2	<0.347	NA
Coumaphos	µg/L (ppb)	8140	NA	—	<0.2	<0.347	NA
Demeton	µg/L (ppb)	8140	NA	—	<0.2	<0.347	NA
Diazinon	µg/L (ppb)	8140	NA	—	<0.2	<0.347	NA
Dichlorvos	µg/L (ppb)	8140	NA	—	<0.2	<0.347	NA
Disulfoton	µg/L (ppb)	8140	NA	—	<0.5	<0.866	NA
Ethoprop	µg/L (ppb)	8140	NA	—	<0.2	<0.347	NA
Fensulfothion	µg/L (ppb)	8140	NA	—	<0.2	<0.347	NA
Fenthion	µg/L (ppb)	8140	NA	—	<0.2	<0.347	NA
Merphos	µg/L (ppb)	8140	NA	—	<0.2	<0.347	NA
Mevinphos	µg/L (ppb)	8140	NA	—	<0.2	<0.347	NA
Naled (Dibrom)	µg/L (ppb)	8140	NA	—	<0.2	<0.347	NA
Parathion Methyl	µg/L (ppb)	8140	NA	—	<0.2	<0.347	NA
Phorate	µg/L (ppb)	8140	NA	—	<0.2	<0.347	NA
Ronnel	µg/L (ppb)	8140	NA	—	<0.2	<0.347	NA
Stirophos	µg/L (ppb)	8140	NA	—	<0.2	<0.347	NA
Tokuthion	µg/L (ppb)	8140	NA	—	<0.2	<0.347	NA
Trichloronate	µg/L (ppb)	8140	NA	—	<0.2	<0.347	NA
Objectives for the Protection of Marine Aquatic Life							
Arsenic	µg/L (ppb)	200.8	670	**2.8**	<2	<4.21	17
Cadmium	µg/L (ppb)	6010/200.7	130	<10	<10	<17.3	88
Chromium (Hexavalent)[i]	µg/L (ppb)	6010/200.7	270	<10	**≈1.9**	<10.8	93
Copper	µg/L (ppb)	6010/200.7	140	<10	**15**	<21.4	690
Lead	µg/L (ppb)	200.8	270	**1.6**	**2.1**	**3.17**	465
Mercury	µg/L (ppb)	7470/245.1	5	<0.2	<0.2	<0.347	1.4
Nickel	µg/L (ppb)	6010/200.7	670	<10	**≈5**	<13.3	142
Selenium	µg/L (ppb)	200.8	2010	**3**	**≈0.94**	**≈3.54**	65
Silver	µg/L (ppb)	6010/200.7	70	<10	<10	<17.3	28
Zinc	µg/L (ppb)	6010/200.7	1620	<50	**56**	<91.5	244
Cyanide, Total	µg/L (ppb)	335.3	130	**22**	**21**	**37.3**	57
Total Residual Chlorine[j]	µg/L (ppb)		270				
Ammonia as N (monthly)	mg/L (ppm)	350.1	80.4	**20** **25** **32** **29** **20** **24**	**33** **29** **28** **27** **27** **24**	NA	NA

[i] Measured as total chromium
[j] Daily measurements are discussed in Chapter 2 of this report

Table C.1 Effluent Monitoring Results

Constituent	Units	Method	Concentration			Mass Emission	
			Permit Limit[a]	January-June[b]	July-December[c]	Loading (kg)	Goal (kg/yr)
NonChlorinated Phenolics	µg/L (ppb)	625	4020	<10	<10	<17.3	NA
Chlorinated Phenolics	µg/L (ppb)	625	130	<10	<10	<17.3	NA
Endosulfan	µg/L (ppb)	608	1.21	<0.005	<0.005	<8.66E-3	3
Endrin	µg/L (ppb)	608	0.27	<0.005	<0.005	<8.66E-3	1
HCH	µg/L (ppb)	608	0.54	<0.005	<0.005	<0.0087	228
Toxicity-Chronic: *M. pyrifera* germination	TUc	CSWRCB	134	**5.6**	**5.6**	NA	NA
Toxicity-Chronic: *M. pyrifera* growth	TUc	CSWRCB	134	**5.6**	**5.6**	NA	NA
Radioactivity Gross α	pCi/L	7110C	15[k]	—	**0.46± 0.89**	NA	NA
Radioactivity Gross β	pCi/L	900.0	50[k]	—	**11.6 ± 3.01**	NA	NA
Objectives for the Protection of Human Health: Non-Carcinogens							
Acrolein	µg/L (ppb)	624	29500	<50	<50	<86.6	NA
Antimony	µg/L (ppb)	6010/200.7	160800	<100	**≈18**	<107.	285
bis-(2-chloroethoxy) methane	µg/L (ppb)	625	590	<2	<2	<3.47	142
bis(2chloro-1methylethyl) ether	µg/L (ppb)	625	160800	<2	<2	<3.47	NA
Chlorobenzene	µg/L (ppb)	624	76400	<0.5	<0.5	<0.866	NA
Chromium III[i]	mg/L (ppm)	6010/200.7	25500	<10	**≈1.9**	<10.8	NA
Di-n-butyl phthalate	µg/L (ppb)	625	469000	<2	<2	<3.47	142
Dichlorobenzene	µg/L (ppb)	624	683000	<0.5	<0.5	<0.866	5.7
1,1-Dichloroethene	µg/L (ppb)	624	951000	<0.5	<0.5	<0.866	3
Diethyl phthalate	µg/L (ppb)	625	4420000	<2	<2	<3.47	191
Dimethyl phthalate	mg/L (ppb)	625	109900	<2	<2	<3.47	142
2-Methyl-4,6-dinitrophenol	µg/L (ppb)	625	29500	<10	<20	<25.4	142
2,4-Dinitrophenol	µg/L (ppb)	625	540	<10	<10	<17.3	342
Ethylbenzene	µg/L (ppb)	624	549000	<0.5	<0.5	<0.866	3
Fluoranthene	µg/L (ppb)	625	2000	<2	<2	<3.47	142
Hexachlorocyclopentadiene	µg/L (ppb)	625	7800	<2	<2	<3.47	NA
Isophorone	µg/L (ppb)	625	20100000	<2	<2	<3.47	142
Nitrobenzene	µg/L (ppb)	625	660	<2	<2	<3.47	142
Thallium	µg/L (ppb)	200.8	1880	<1	**≈0.067**	<0.980	285
Toluene	µg/L (ppb)	624	11400000	<0.5	**≈0.31**	<0.713	4
1,1,2,2-Tetrachloroethane	µg/L (ppb)	624	160800	<0.5	<0.5	<0.866	3
Tributyltin[l]	µg/L (ppb)	—	0.19	—	—	NA	NA
1,1,1-Trichloroethane	µg/L (ppb)	624	72400000	<0.5	<0.5	<0.866	3
1,1,2-Trichloroethane	µg/L (ppb)	624	5760000	<0.5	<0.5	<0.866	3
Objectives for the Protection of Human Health: Carcinogens							
Acrylonitrile	µg/L (ppb)	624	13.4	<20	<20	<34.7	NA
Aldrin	µg/L (ppb)	608	0.003	<0.005	<0.005	<8.66E-3	0.01[m]
Benzene	µg/L (ppb)	624	790	<0.5	<0.5	<0.866	3
Benzidine	µg/L (ppb)	625	0.009	<20	<20	**<34.7[n]**	0.03[m]
Beryllium	µg/L (ppb)	200.7/6010	4.42	<10	<10	<17.3	28[m]
bis-(2-chloroethyl) ether	µg/L (ppb)	625	6.03	<2	<2	<3.47	17[m]
bis-(2-ethylhexyl) phthalate	µg/L (ppb)	625	470	**6.6**	**10**	**14.2**	320

k Drinking-water limit
l Tributyltin analysis is not required under the current Monitoring and Reporting Program
m Mass-emission goal computed from permit limit because historically, the limit has been below the PQL
n Comparison of loading and emission goal is indeterminate because maximum loading computed from PQL exceeds the emission goal

Table C.1 Effluent Monitoring Results

			Concentration			Mass Emission	
Constituent	**Units**	**Method**	**Permit Limit**[a]	**January-June**[b]	**July-December**[c]	**Loading (kg)**	**Goal (kg/yr)**
Carbon Tetrachloride	µg/L (ppb)	624	120	<0.5	<0.5	<0.866	3
Chlordane	µg/L (ppb)	608	0.0031	<1	<0.5	**<1.33**[n]	0.0088[m]
Chloroform	µg/L (ppb)	624	17400	<0.5	**≈0.30**	<0.705	5
DDT	µg/L (ppb)	608	0.0228	<0.005	<0.005	<8.66E-3	0.06
1,4-Dichlorobenzene	µg/L (ppb)	624	2410	<0.5	**≈0.22**	<0.641	57
3,3-Dichlorobenzidine	µg/L (ppb)	625	1.09	<10	<10	**<17.3**[n]	3.1[m]
1,2-Dichloroethane	µg/L (ppb)	624	17400	<0.5	<0.5	<0.866	3
Methylene chloride	µg/L (ppb)	624	60300	<1	**≈0.16**	<1.06	NA
1,3-Dichloropropene	µg/L (ppb)	624	1190	<0.5	<0.5	<0.866	NA
Dieldrin	µg/L (ppb)	608	0.0054	<0.005	<0.005	<8.66E-3	0.02[m]
2,4-Dinitrotoluene	µg/L (ppb)	625	350	<2	<2	<3.47	142
1,2-Diphenylhydrazine	µg/L (ppb)	625	21	<2	<2	<3.47	60[m]
Halomethanes	µg/L (ppb)	624	17400	<1	<1	<1.73	25
Heptachlor	µg/L (ppb)	608	0.0965	<0.005	<0.005	<8.66E-3	0.27[m]
Hexachlorobenzene	µg/L (ppb)	625	0.0281	<2	<2	**<3.47**[n]	0.08[m]
Hexachlorobutadiene	µg/L (ppb)	625	1880	<2	<2	<3.47	142
Hexachloroethane	µg/L (ppb)	625	340	<2	<2	<3.47	142
N-Nitrosodimethylamine	µg/L (ppb)	625	980	<2	<2	<3.47	342
N-Nitrosodiphenylamine	µg/L (ppb)	625	340	<2	<2	<3.47	142
PAHs	µg/L (ppb)	625	1.18	<2	<3	**<4.27**[n]	3.4[m]
PCBs	µg/L (ppb)	608	0.0026	<0.2	<0.2	**<0.347**[n]	0.0073[m]
Dibenzofuran	µg/L (ppb)	625	NA	<2	<2	<3.47	57
Dioxin (Total TCDD)	pg/L	1613	0.52	—	**0.159**	**2.75E-7**	1.42E-06[m]
Tetrachloroethene	µg/L (ppb)	624	13300	<0.5	<0.5	<0.866	4
Toxaphene	µg/L (ppb)	608	0.0281	<2	<2	**<3.47**[n]	0.08[m]
Trichloroethene (TCE)	µg/L (ppb)	624	3620	<0.5	<0.5	<0.866	3
2,4,6-Trichlorophenol	µg/L (ppb)	625	40	<5	<5	<8.66	114[m]
Vinyl chloride	µg/L (ppb)	624	4820	<0.5	<0.5	<0.866	3

Table C.2 Biosolids Monitoring Results

Constituent	Units[o]	Method Number	Concentration[p]
Percent Solids	%	160.3	**90**
Oil and Grease	mg/Kg (ppm)	1664	**39000**
Inorganic Parameters			
pH	pH units	9045	**6.9**
Cyanide, Total	mg/Kg (ppm)	9010/9014	<0.1[q]
Nitrogen as NO_3	mg/Kg (ppm)	300.0	**69**
Nitrogen, Total Kjeldahl (TKN)	mg/Kg (ppm)	351.3	**23000**
Ammonia as N	mg/Kg (ppm)	350.2	**2000**
Organic Nitrogen[r]	mg/Kg (ppm)	—	**21000**
Phosphate, Total	mg/Kg (ppm)	365.2	**26000**
Total Dissolved Solids STLC[s]	mg/L (ppm)	160.1	**3000**
Metals			
Antimony	mg/Kg (ppm)	6010	<3
Arsenic	mg/Kg (ppm)	6010	**11**
Barium	mg/Kg (ppm)	6010	**570**
Beryllium	mg/Kg (ppm)	6010	<0.2
Boron	mg/Kg (ppm)	6010	**14**
Cadmium	mg/Kg (ppm)	6010	**5.5**
Chromium	mg/Kg (ppm)	6010	**39**
Cobalt	mg/Kg (ppm)	6010	**4.5**
Copper	mg/Kg (ppm)	6010	**550**
Copper STLC Waste Extraction	mg/L (ppm)	6010	**10**
Lead	mg/Kg (ppm)	6010	**56**
Lead STLC Waste Extraction	mg/L (ppm)	6010	**3.6**[t]
Mercury	mg/Kg (ppm)	7471	**1.3**
Molybdenum	mg/Kg (ppm)	6010	**18**
Nickel	mg/Kg (ppm)	6010	**37**
Selenium	mg/Kg (ppm)	6010	**31**
Selenium STLC Waste Extraction	mg/L (ppm)	6020	<0.2
Silver	mg/Kg (ppm)	6010	**4.3**
Thallium	mg/Kg (ppm)	6010	<3
Vanadium	mg/Kg (ppm)	6010	**21**
Zinc	mg/Kg (ppm)	6010	**1300**
Organochlorine Pesticides and PCBs			
Aldrin	mg/Kg (ppm)	8081A	<0.002[u]
Alpha-BHC	mg/Kg (ppm)	8081A	<0.002
Beta-BHC	mg/Kg (ppm)	8081A	<0.002

[o] Bulk (wet) weight concentration
[p] Sample collected on 8 September 2005
[q] Except where otherwise indicated, for biosolids chemical analyses, "<" indicates the substance was not detected above the indicated detection limit for purposes of reporting (DLR)
[r] Organic Nitrogen is a Total Kjeldahl Nitrogen minus Nitrogen as Ammonia
[s] Modified STLC extraction
[t] Average of results from two separate extractions of the same biosolids sample
[u] For the results reported for methods 8081A and 8082, "<" indicates the substance was not detected above the indicated PQL

Table C.2 Biosolids Monitoring Results

Constituent	Units[o]	Method Number	Concentration[p]
Gamma-Chlordane	mg/Kg (ppm)	8081A	<0.002
Alpha-Chlordane	mg/Kg (ppm)	8081A	<0.002
4,4'-DDD	mg/Kg (ppm)	8081A	<0.004
4,4'-DDE	mg/Kg (ppm)	8081A	<0.004
4,4'-DDT	mg/Kg (ppm)	8081A	<0.004
Delta-BHC	mg/Kg (ppm)	8081A	<0.002
Dieldrin	mg/Kg (ppm)	8081A	<0.004
Endosulfan I	mg/Kg (ppm)	8081A	<0.002
Endosulfan II	mg/Kg (ppm)	8081A	<0.004
Endosulfan Sulfate	mg/Kg (ppm)	8081A	<0.004
Endrin	mg/Kg (ppm)	8081A	<0.004
Endrin Aldehyde	mg/Kg (ppm)	8081A	<0.004
Endrin Ketone	mg/Kg (ppm)	8081A	<0.004
Gamma-BHC (Lindane)	mg/Kg (ppm)	8081A	<0.002
Heptachlor	mg/Kg (ppm)	8081A	<0.002
Heptachlor Epoxide	mg/Kg (ppm)	8081A	<0.002
Methoxychlor	mg/Kg (ppm)	8081A	<0.017
Toxaphene	mg/Kg (ppm)	8081A	<0.170
PCB-1016	mg/Kg (ppm)	8082	<0.033
PCB-1221	mg/Kg (ppm)	8082	<0.067
PCB-1232	mg/Kg (ppm)	8082	<0.033
PCB-1242	mg/Kg (ppm)	8082	<0.033
PCB-1248	mg/Kg (ppm)	8082	<0.033
PCB-1254	mg/Kg (ppm)	8082	<0.033
PCB-1260	mg/Kg (ppm)	8082	<0.033
Organophosphorus Pesticides			
Azinphos Methyl	mg/Kg (ppm)	8141A	<25[v]
Bolstar	mg/Kg (ppm)	8141A	<25
Bensulide	mg/Kg (ppm)	8141A	<25
Carbofenthion (Trithion)	mg/Kg (ppm)	8141A	<100
Chlorfenvinphos (Supona)	mg/Kg (ppm)	8141A	<25
Chlorpyrifos (Dursban)	mg/Kg (ppm)	8141A	<15
Chlorpyrifos-methyl	mg/Kg (ppm)	8141A	<15
Ciodrin (Crotophos)	mg/Kg (ppm)	8141A	<25
Coumaphos (Co-Ral)	mg/Kg (ppm)	8141A	<75
DEF	mg/Kg (ppm)	8141A	<25
Demeton (Systox) O/S Analogues	mg/Kg (ppm)	8141A	<25
Diazinon	mg/Kg (ppm)	8141A	<25
Dibrom (Naled)	mg/Kg (ppm)	8141A	<25
Dicrotophos (Didrin)	mg/Kg (ppm)	8141A	<25
Dimethoate (Cygon)	mg/Kg (ppm)	8141A	<25
Disulfoton (Disyston)	mg/Kg (ppm)	8141A	<15
EPN	mg/Kg (ppm)	8141A	<50
Ethion	mg/Kg (ppm)	8141A	<25
Ethoprop (Modap)	mg/Kg (ppm)	8141A	<25
Fenamiphos (Nemacur)	mg/Kg (ppm)	8141A	<25
Fenitrothioin (Sumithion)	mg/Kg (ppm)	8141A	<25

[v] For the results reported for method 8141A, "<" indicates the substance was not detected above the indicated MDL

Table C.2 Biosolids Monitoring Results

Constituent	Units[o]	Method Number	Concentration[p]
Fenthion (Baytex)	mg/Kg (ppm)	8141A	<25
Fonofos (Dyfonate)	mg/Kg (ppm)	8141A	<25
Imidan (Phosmet)	mg/Kg (ppm)	8141A	<25
Isofenphos (Oftanol)	mg/Kg (ppm)	8141A	<25
Malathion	mg/Kg (ppm)	8141A	<25
Methidathion (Supracide)	mg/Kg (ppm)	8141A	<25
Methyl Parathion	mg/Kg (ppm)	8141A	<25
Mevinphos (Phosdrin)	mg/Kg (ppm)	8141A	<25
Parathion	mg/Kg (ppm)	8141A	<25
Phorate (Thimet)	mg/Kg (ppm)	8141A	<25
Phosalone (Zolone)	mg/Kg (ppm)	8141A	<75
Phosphamidon (Dimecron)	mg/Kg (ppm)	8141A	<50
Primiphos-methyl	mg/Kg (ppm)	8141A	<25
Profenofos (Curacron)	mg/Kg (ppm)	8141A	<50
Propetaminphos (Safrontin)	mg/Kg (ppm)	8141A	<25
Ronnel (Fenchlorfos)	mg/Kg (ppm)	8141A	<25
Tetrachlorvinphos (Gardona)	mg/Kg (ppm)	8141A	<25
Thionazin (Zinophos)	mg/Kg (ppm)	8141A	<25
Acetamaprid	mg/Kg (ppm)	8141A	<100
Ametryn	mg/Kg (ppm)	8141A	<50
Atrazine	mg/Kg (ppm)	8141A	<25
Azoxystrobin	mg/Kg (ppm)	8141A	<25
Benthiocarb	mg/Kg (ppm)	8141A	<100
Cyanazine (Bladex)	mg/Kg (ppm)	8141A	<25
Diphenyl Amine	mg/Kg (ppm)	8141A	<100
Hexazinone (Velpar)	mg/Kg (ppm)	8141A	<50
Imazalil	mg/Kg (ppm)	8141A	<100
Metalaxyl (Ridomil)	mg/Kg (ppm)	8141A	<100
Metolachlor (Dual)	mg/Kg (ppm)	8141A	<50
Metribuzin (Sencor)	mg/Kg (ppm)	8141A	<50
Molinate (Ordram)	mg/Kg (ppm)	8141A	<50
Myclobutanil	mg/Kg (ppm)	8141A	<25
Prometon	mg/Kg (ppm)	8141A	<25
Prometryne	mg/Kg (ppm)	8141A	<25
Pymetrazine	mg/Kg (ppm)	8141A	<25
Simazine	mg/Kg (ppm)	8141A	<25
Terbacil	mg/Kg (ppm)	8141A	<250
Thiabandazole	mg/Kg (ppm)	8141A	<50
Volatile Organic Compounds			
Benzene	mg/Kg (ppm)	8260	<0.005
Bromobenzene	mg/Kg (ppm)	8260	<0.005
Bromochloromethane	mg/Kg (ppm)	8260	<0.005
Bromodichloromethane	mg/Kg (ppm)	8260	<0.005
Bromoform	mg/Kg (ppm)	8260	<0.005
Bromomethane	mg/Kg (ppm)	8260	<0.005
t-Butylbenzene	mg/Kg (ppm)	8260	<0.005
n-Butylbenzene	mg/Kg (ppm)	8260	<0.005
sec-Butylbenzene	mg/Kg (ppm)	8260	<0.005
Carbon Tetrachloride	mg/Kg (ppm)	8260	<0.005
Chlorobenzene	mg/Kg (ppm)	8260	<0.005

Table C.2 Biosolids Monitoring Results

Constituent	Units[o]	Method Number	Concentration[p]
Chloroethane	mg/Kg (ppm)	8260	<0.005
2 Chloroethylvinyl ether	mg/Kg (ppm)	8260	<0.005
Chloroform	mg/Kg (ppm)	8260	<0.005
Chloromethane	mg/Kg (ppm)	8260	<0.005
2-Chlorotoluene	mg/Kg (ppm)	8260	<0.005
4-Chlorotoluene	mg/Kg (ppm)	8260	<0.005
1,2-Dibromo-3-Chloropropane	mg/Kg (ppm)	8260	<0.005
Dibromochloromethane	mg/Kg (ppm)	8260	<0.005
Dibromomethane	mg/Kg (ppm)	8260	<0.005
1,2-Dibromoethane	mg/Kg (ppm)	8260	<0.005
Dichlorodifluourmethane	mg/Kg (ppm)	8260	<0.005
1,2-Dichlorobenzene	mg/Kg (ppm)	8260	<0.005
1,3-Dichlorobenzene	mg/Kg (ppm)	8260	<0.005
1,4-Dichlorobenzene	mg/Kg (ppm)	8260	<0.005
1,1-Dichloroethane	mg/Kg (ppm)	8260	<0.005
1,2-Dichloroethane	mg/Kg (ppm)	8260	<0.005
1,1-Dichloroethene	mg/Kg (ppm)	8260	<0.005
cis-1,3-Dichloropropene	mg/Kg (ppm)	8260	<0.005
trans-1,2-Dichloroethene	mg/Kg (ppm)	8260	<0.005
1,2-Dichloropropane	mg/Kg (ppm)	8260	<0.005
1,3-Dichloropropane	mg/Kg (ppm)	8260	<0.005
2,2-Dichloropropane	mg/Kg (ppm)	8260	<0.005
1,1-Dichloropropene	mg/Kg (ppm)	8260	<0.005
cis-1,3-Dichloropropene	mg/Kg (ppm)	8260	<0.005
trans-1,3-Dichloropropene	mg/Kg (ppm)	8260	<0.005
Ethyl Benzene	mg/Kg (ppm)	8260	<0.005
Hexachlorobutadiene	mg/Kg (ppm)	8260	<0.005
Isopropylbenzene	mg/Kg (ppm)	8260	<0.005
4-Isopropyltoluene	mg/Kg (ppm)	8260	<0.005
Methylene chloride	mg/Kg (ppm)	8260	<0.01
MTBE	mg/Kg (ppm)	8260	<0.005
MTBE	mg/Kg (ppm)	8260	<0.005
Napthalene	mg/Kg (ppm)	8260	<0.005
n-Propylbenzene	mg/Kg (ppm)	8260	<0.005
Styrene	mg/Kg (ppm)	8260	<0.005
1,1,1,2-Tetrachloroethane	mg/Kg (ppm)	8260	<0.005
1,1,2,2-Tetrachloroethane	mg/Kg (ppm)	8260	<0.005
Tetrachloroethene	mg/Kg (ppm)	8260	<0.005
Toluene	mg/Kg (ppm)	8260	<0.005
1,2,3-Trichlorobenzene	mg/Kg (ppm)	8260	<0.005
1,2,4-Trichlorobenzene	mg/Kg (ppm)	8260	<0.005
1,1,1-Trichloroethane	mg/Kg (ppm)	8260	<0.005
1,1,2-Trichloroethane	mg/Kg (ppm)	8260	<0.005
Trichloroethene	mg/Kg (ppm)	8260	<0.005
Trichlorofluoromethane	mg/Kg (ppm)	8260	<0.005
1,2,3-Trichloropropane	mg/Kg (ppm)	8260	<0.005
1,2,4-Trimethylbenzene	mg/Kg (ppm)	8260	<0.005
1,3,5-Trimethylbenzene	mg/Kg (ppm)	8260	<0.005
Vinyl Chloride	mg/Kg (ppm)	8260	<0.005
m,p-Xylene	mg/Kg (ppm)	8260	<0.005

Table C.2 Biosolids Monitoring Results

Constituent	Units[o]	Method Number	Concentration[p]
o-Xylene	mg/Kg (ppm)	8260	<0.005
Base Neutral and Acid Organic Compounds			
Acenaphthene	mg/Kg (ppm)	8270	<10
Acenaphthylene	mg/Kg (ppm)	8270	<10
Anthracene	mg/Kg (ppm)	8270	<10
Azobenzene	mg/Kg (ppm)	8270	<10
Benzo(a)anthracene	mg/Kg (ppm)	8270	<10
Benzo(a)pyrene	mg/Kg (ppm)	8270	<10
Benzo(b)fluoranthene	mg/Kg (ppm)	8270	<10
Benzo(g,h,i)perylene	mg/Kg (ppm)	8270	<10
Benzo(k)fluoranthene	mg/Kg (ppm)	8270	<10
Benzoic Acid	mg/Kg (ppm)	8270	<10
Benzyl butyl phthalate	mg/Kg (ppm)	8270	<10
bis(2-chloroethoxy) methane	mg/Kg (ppm)	8270	<10
bis(2-chloroethyl) ether	mg/Kg (ppm)	8270	<10
bis(2-chloroisopropyl) ether	mg/Kg (ppm)	8270	<10
bis(2-ethylhexyl) phthalate	mg/Kg (ppm)	8270	**36**
4-Bromodiphenyl ether	mg/Kg (ppm)	8270	<10
Carbazole	mg/Kg (ppm)	8270	<10
4-Chloro-3methylphenol	mg/Kg (ppm)	8270	<10
4-Chloroaniline	mg/Kg (ppm)	8270	<10
2-Chloronaphthalene	mg/Kg (ppm)	8270	<10
2-Chlorophenol	mg/Kg (ppm)	8270	<10
4-Chlorophenyl phenyl ether	mg/Kg (ppm)	8270	<10
Chrysene	mg/Kg (ppm)	8270	<10
Dibenzo(a,h)anthracene	mg/Kg (ppm)	8270	<10
Dibenzofuran	mg/Kg (ppm)	8270	<10
Dibutyl phthalate	mg/Kg (ppm)	8270	<10
1,2-Dichlorbenzene	mg/Kg (ppm)	8270	<10
1,3-Dichlorbenzene	mg/Kg (ppm)	8270	<10
1,4-Dichlorbenzene	mg/Kg (ppm)	8270	<10
3,3-Dichlorobenzidine	mg/Kg (ppm)	8270	<10
2,4-Dichlorophenol	mg/Kg (ppm)	8270	<10
Diethyl phthalate	mg/Kg (ppm)	8270	<10
Dimethyl phthalate	mg/Kg (ppm)	8270	<10
2,4-Dimethylphenol	mg/Kg (ppm)	8270	<10
2,4-Dinitrophenol	mg/Kg (ppm)	8270	<10
2,4-Dinitrotoluene	mg/Kg (ppm)	8270	<10
2,6-Dinitrotoluene	mg/Kg (ppm)	8270	<10
Di-n-octylphthalate	mg/Kg (ppm)	8270	<10
Fluoranthene	mg/Kg (ppm)	8270	<10
Fluorene	mg/Kg (ppm)	8270	<10
Hexachloroethane	mg/Kg (ppm)	8270	<10
Hexachlorobenzene	mg/Kg (ppm)	8270	<10
Hexachlorobutadiene	mg/Kg (ppm)	8270	<10
Hexachlorocyclopentadiene	mg/Kg (ppm)	8270	<10
Indeno(1,2,3-c,d)pyrene	mg/Kg (ppm)	8270	<10
Isophorone	mg/Kg (ppm)	8270	<10
2-Methyl-4,6-dinitrophenol	mg/Kg (ppm)	8270	<10
2-Methylnaphthalene	mg/Kg (ppm)	8270	<10

Table C.2 Biosolids Monitoring Results

Constituent	Units[o]	Method Number	Concentration[p]
2-Methylphenol	mg/Kg (ppm)	8270	<10
4-Methylphenol	mg/Kg (ppm)	8270	<10
Naphthalene	mg/Kg (ppm)	8270	<10
2-Nitroaniline	mg/Kg (ppm)	8270	<10
3-Nitroaniline	mg/Kg (ppm)	8270	<10
4-Nitroaniline	mg/Kg (ppm)	8270	<10
Nitrobenzene	mg/Kg (ppm)	8270	<10
2-Nitrophenol	mg/Kg (ppm)	8270	<10
4-Nitrophenol	mg/Kg (ppm)	8270	<10
N-Nitrosodimethylamine	mg/Kg (ppm)	8270	<10
N-Nitrosodiphenylamine	mg/Kg (ppm)	8270	<10
Pentachlorophenol	mg/Kg (ppm)	8270	<10
Phenanthrene	mg/Kg (ppm)	8270	<10
Phenol	mg/Kg (ppm)	8270	<10
Pyrene	mg/Kg (ppm)	8270	<10
1,2,4-Trichlorobenzene	mg/Kg (ppm)	8270	<10
2,4,5-Trichlorophenol	mg/Kg (ppm)	8270	<10
2,4,6-Trichlorophenol	mg/Kg (ppm)	8270	<10
Other Parameters			
Asbestos	%	600/R-93/116	ND[w]

[w] None detected

APPENDIX D

Sediment Physicochemistry: October 2005 Survey

Table D.1 Description of the Sediment Grab Samples Collected in October 2005 aboard the F/V *Bonnie Marietta*

Station	Sample	Date	Time[a]	Penetration Depth (cm)	Latitude	Longitude	Sediment Type	Sediment Color	Comments/ Observations[b]
1	Bio 1	10/04/05	06:57:06	9.0	35° 23.716' N	120° 52.672' W	Sand	Lt. Brown	
	Bio 2	10/04/05	07:00:41	8.0	35° 23.717' N	120° 52.671' W	Sand	Lt. Brown	
	Bio 3	10/04/05	07:03:22	9.0	35° 23.718' N	120° 52.671' W	Sand	Lt. Brown	
	Bio 4	10/04/05	07:08:57	7.0	35° 23.716' N	120° 52.680' W	Sand	Lt. Brown	
	Bio 5	10/04/05	07:11:42	7.5	35° 23.715' N	120° 52.679' W	Sand	Lt. Brown	
	Chem 1	10/05/05	07:13:14	N/A[c]	35° 23.725' N	120° 52.672' W	Sand	Lt. Brown	Sediment Chemistry
	Chem 2	10/05/05	07:25:42	N/A	35° 23.727' N	120° 52.679' W	Sand	Lt. Brown	Dissolved Sulfide
2	Bio 1	10/04/05	07:45:18	7.0	35° 23.277' N	120° 52.505' W	Sand	Lt. Brown	One Midshipman Fish
	Bio 2	10/04/05	07:47:50	9.0	35° 23.277' N	120° 52.505' W	Sand	Lt. Brown	
	Bio 3	10/04/05	07:50:19	7.0	35° 23.276' N	120° 52.504' W	Sand	Lt. Brown	
	Bio 4	10/04/05	07:52:59	7.5	35° 23.274' N	120° 52.505' W	Sand	Lt. Brown	
	Bio 5	10/04/05	07:55:43	7.5	35° 23.273' N	120° 52.507' W	Sand	Lt. Brown	
	Chem 1	10/05/05	07:37:15	N/A	35° 23.278' N	120° 52.508' W	Sand	Lt. Brown	
3	Bio 1	10/04/05	09:05:45	7.5	35° 23.228' N	120° 52.499' W	Sand	Lt. Brown	
	Bio 2	10/04/05	09:10:45	7.5	35° 23.228' N	120° 52.505' W	Sand	Lt. Brown	
	Bio 3	10/04/05	09:15:31	7.5	35° 23.229' N	120° 52.505' W	Sand	Lt. Brown	
	Bio 4	10/04/05	09:18:03	7.5	35° 23.228' N	120° 52.503' W	Sand	Lt. Brown	
	Bio 5	10/04/05	09:22:50	8.5	35° 23.229' N	120° 52.509' W	Sand	Lt. Brown	
	Chem 1	10/05/05	07:53:54	N/A	35° 23.228' N	120° 52.504' W	Sand	Lt. Brown	
4	Bio 1	10/04/05	08:26:45	7.5	35° 23.210' N	120° 52.503' W	Sand	Lt. Brown	
	Bio 2	10/04/05	08:34:13	6.5	35° 23.207' N	120° 52.505' W	Sand	Lt. Brown	Low volume
	Bio 3	10/04/05	08:37:07	7.0	35° 23.208' N	120° 52.512' W	Sand	Lt. Brown	One Midshipman Fish
	Bio 4	10/04/05	08:43:11	7.5	35° 23.210' N	120° 52.512' W	Sand	Lt. Brown	
	Bio 5	10/04/05	08:45:48	—[d]	35° 23.211' N	120° 52.514' W	Sand	Lt. Brown	
	Chem 1	10/05/05	08:09:39	N/A	35° 23.208' N	120° 52.507' W	Sand	Lt. Brown	
5	Bio 1	10/04/05	09:56:05	7.0	35° 23.187' N	120° 52.502' W	Sand	Lt. Brown	
	Bio 2	10/04/05	10:05:58	7.0	35° 23.186' N	120° 52.509' W	Sand	Lt. Brown	
	Bio 3	10/04/05	10:21:24	7.0	35° 23.186' N	120° 52.508' W	Sand	Lt. Brown	
	Bio 4	10/04/05	10:23:39	7.0	35° 23.185' N	120° 52.507' W	Sand	Lt. Brown	
	Bio 5	10/04/05	10:25:57	7.0	35° 23.186' N	120° 52.510' W	Sand	Lt. Brown	
	Chem 1	10/05/05	08:24:35	N/A	35° 23.190' N	120° 52.511' W	Sand	Lt. Brown	

[a] Time when the sediment grab was collected (trip time of the grab).
[b] No odors emanating from the sediment samples were noted at any station.
[c] Penetration depth is not pertinent for grab samples collected for physiochemical analysis.
[d] Penetration depth was not recorded.

Table D.1 Description of the Sediment Grab Samples Collected in October 2005 aboard the F/V *Bonnie Marietta*

Station	Sample	Date	Time[a]	Penetration Depth (cm)	Latitude	Longitude	Sediment Type	Sediment Color	Comments/ Observations[b]
6	Bio 1	10/04/05	10:46:14	8.0	35° 23.165' N	120° 52.502' W	Sand	Lt. Brown	
	Bio 2	10/04/05	10:48:24	7.5	35° 23.167' N	120° 52.501' W	Sand	Lt. Brown	
	Bio 3	10/04/05	10:52:47	8.5	35° 23.169' N	120° 52.497' W	Sand	Lt. Brown	
	Bio 4	10/04/05	10:59:41	7.0	35° 23.173' N	120° 52.496' W	Sand	Lt. Brown	
	Bio 5	10/04/05	11:04:32	9.0	35° 23.174' N	120° 52.495' W	Sand	Lt. Brown	
	Chem 1	10/05/05	08:38:24	N/A	35° 23.166' N	120° 52.503' W	Sand	Lt. Brown	
7	Bio 1	10/04/05	11:22:31	8.0	35° 23.114' N	120° 52.502' W	Sand	Lt. Brown	
	Bio 2	10/04/05	11:32:07	7.5	35° 23.111' N	120° 52.504' W	Sand	Lt. Brown	
	Bio 3	10/04/05	11:37:07	8.0	35° 23.108' N	120° 52.506' W	Sand	Lt. Brown	
	Bio 4	10/04/05	11:43:48	8.0	35° 23.106' N	120° 52.507' W	Sand	Lt. Brown	
	Bio 5	10/04/05	11:45:58	7.0	35° 23.105' N	120° 52.507' W	Sand	Lt. Brown	
	Chem 1	10/05/05	09:01:00	N/A	35° 23.118' N	120° 52.504' W	Sand	Lt. Brown	
8	Bio 1	10/04/05	12:00:00	8.0	35° 23.201' N	120° 52.549' W	Sand	Lt. Brown	
	Bio 2	10/04/05	12:02:24	7.0	35° 23.199' N	120° 52.547' W	Sand	Lt. Brown	
	Bio 3	10/04/05	12:04:41	7.0	35° 23.198' N	120° 52.548' W	Sand	Lt. Brown	
	Bio 4	10/04/05	12:07:02	7.5	35° 23.198' N	120° 52.548' W	Sand	Lt. Brown	
	Bio 5	10/04/05	12:09:19	7.5	35° 23.197' N	120° 52.547' W	Sand	Lt. Brown	
	Chem 1	10/05/05	09:16:13	N/A	35° 23.195' N	120° 52.546' W	Sand	Lt. Brown	
9	Bio 1	10/04/05	12:20:23	8.0	35° 23.195' N	120° 52.456' W	Sand	Lt. Brown	
	Bio 2	10/04/05	13:54:03	9.0	35° 23.195' N	120° 52.468' W	Sand	Lt. Brown	
	Bio 3	10/04/05	13:56:13	9.5	35° 23.195' N	120° 52.467' W	Sand	Lt. Brown	
	Bio 4	10/04/05	13:58:36	7.0	35° 23.194' N	120° 52.468' W	Sand	Lt. Brown	
	Bio 5	10/04/05	14:00:48	7.0	35° 23.194' N	120° 52.467' W	Sand	Lt. Brown	
	Chem 1	10/05/05	09:08:50	N/A	35° 23.187' N	120° 52.463' W	Sand	Lt. Brown	

Table D.2 Summary of Analytical Methods Used for Physicochemical Determinations of Sediment

Parameter	Method	Description	Reference
Biochemical Oxygen Demand 5-Day	5210B	Dissolved Oxygen Probe	6
Total Kjeldahl Nitrogen (TKN)	351.2	Block Digestion/Colorimetry	1
Dissolved Sulfides	376.2	Spectrophotometric	1
Total Volatile Solids	160.4	Dry ashing/Gravimetry	1
Moisture	160.3	Evaporation/Gravimetry	1
Sediment Grain Size	MRS, 1998	Sieve/Pipette without salt	2,3,4
Aluminum	6010	ICP	5
Arsenic	7060	GFAAS	5
Cadmium	6010	ICP	5
Chromium	6010	ICP	5
Copper	6010	ICP	5
Iron	6010	ICP	5
Lead	7421	CVAAS	5
Mercury	7471	CVAAS	5
Nickel	6010	ICP	5
Silver	272.2	GFAAS	1
Zinc	6010	ICP	5
Organochlorine Pesticides	8080	GC	5
PCBs	8080	GC	5
Phenolic Compounds	8270	GC/MS	5
Oil and Grease	1664	Extraction/Gravimetry	7

CVAAS = Cold Vapor Atomic Absorption Spectrometry
ICP/MS = Inductively Coupled Plasma/Mass Spectrometry
GC = Gas Chromatography
GC/MS = Gas Chromatography/Mass Spectrometry
GFAAS = Graphite Furnace Atomic Absorption Spectrometry

References:

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4. Coats, D.A., Imamura, E., Fukuyama, A.K., Skalski, J.R., Kimura, S., and J. Steinbeck. 1998. Monitoring of Biological Recovery of Prince William Sound Intertidal Sites Impacted by the *Exxon Valdez* Oil Spill.1997 Biological Monitoring Survey. NOAA Technical Memorandum NOS ORCA NOS OR&R 1.
5. U.S. Environmental Protection Agency. 1986. Test methods for evaluating solid wastes physical/chemical methods. EPA SW-846. 3rd ed., revised.
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7. U.S. Environmental Protection Agency. 1999. Method 1664 Revision A: N-Hexane Extractable Material (HEM; Oil and Grease) and Silica Gel Treated N-Hexane Extractable Material (SGT-HEM; Non-polar Material) by Extraction and Gravimetry. February 1999. EPA-821-R-98-002.

Table D.3 Grain-Size Distribution in Samples Collected during the October 2005 Benthic Survey

Phi	-2 to -1	-1 to 0	0 to 1	1 to 2	2 to 3	3 to 4	4 to 9	>9
Size (µm)	2000	1000	500	250	125	62.5	62.5 to 2	<2
Class	Gravel	Vry Crs Snd	Crs Sand	Med Snd	Fine Snd	Vry Fine Snd	Silt	Clay
Station 1	0.109%	0.039%	0.064%	1.508%	68.671%	28.698%	0.496%	0.415%
Station 2	0.041%	0.044%	0.419%	1.877%	81.183%	15.756%	0.333%	0.345%
Station 3	0.000%	0.010%	0.517%	2.246%	81.059%	15.512%	0.320%	0.337%
Station 4	0.019%	0.057%	0.269%	1.789%	84.968%	12.274%	0.284%	0.340%
Station 4 (dup)	0.000%	0.025%	0.462%	2.809%	85.867%	10.199%	0.220%	0.418%
Station 5	0.011%	0.037%	0.455%	2.139%	82.269%	14.448%	0.266%	0.376%
Station 6	0.000%	0.008%	0.379%	1.506%	80.205%	17.299%	0.314%	0.289%
Station 7	0.000%	0.017%	0.575%	2.043%	80.916%	15.684%	0.378%	0.386%
Station 8	0.000%	0.029%	0.429%	1.324%	78.225%	19.147%	0.527%	0.320%
Station 9	0.000%	0.026%	0.457%	1.978%	83.296%	13.637%	0.284%	0.321%
Station 9 (dup)	0.000%	0.008%	0.067%	3.078%	79.293%	16.704%	0.513%	0.337%

	Dry Weight (grams)			Percentage Summary		
Phi	<-1 to 4	>4		-2 to -1	-1 to 4	4 to >9
Size (µm)	2000 to 62.5	<62.5		2000	1000 to 62.5	<62.5
Class	Coarse	Fine	Total	Gravel	Sand	Mud
Station 1	135.865	1.249	**137.114**	0.109%	98.980%	0.911%
Station 2	131.196	0.896	**132.092**	0.041%	99.280%	0.678%
Station 3	124.170	0.821	**124.991**	0.000%	99.343%	0.657%
Station 4	125.901	0.791	**126.692**	0.019%	99.357%	0.624%
Station 4 (dup)	126.455	0.812	**127.267**	0.000%	99.362%	0.638%
Station 5	119.586	0.772	**120.358**	0.011%	99.348%	0.641%
Station 6	126.749	0.769	**127.518**	0.000%	99.397%	0.603%
Station 7	126.111	0.971	**127.082**	0.000%	99.236%	0.764%
Station 8	120.390	1.028	**121.418**	0.000%	99.153%	0.847%
Station 9	126.010	0.767	**126.777**	0.000%	99.395%	0.605%
Station 9 (dup)	108.315	0.928	**109.243**	0.000%	99.151%	0.849%

Table D.4 Grain-Size Statistics for the October 2005 Survey

Station	Mean (ϕ)	σ (ϕ)	Skewness (ϕ)	Mean (μm)	Percent Coarser than Listed Diameter (μm)				
					5%	16%	50%	84%	95%
1	2.82	0.381	0.10	142	213	181	143	109	87
2	2.69	0.334	0.11	155	222	191	156	124	98
3	2.69	0.337	0.10	155	226	192	157	125	98
4	2.66	0.309	0.11	158	221	192	160	130	103
4dup	2.61	0.314	0.08	164	233	199	164	134	107
5	2.67	0.331	0.10	157	225	193	158	126	100
6	2.72	0.325	0.11	152	215	186	153	122	97
7	2.69	0.337	0.11	155	224	191	157	124	97
8	2.74	0.338	0.14	150	213	185	152	119	93
9	2.67	0.320	0.10	157	223	192	159	128	101
9dup	2.67	0.371	0.11	157	234	198	159	122	95

APPENDIX E

Infaunal Biology

INFAUNAL INDICES

1.1 SPECIES DIVERSITY

The Shannon diversity index (*H′*) (Shannon and Weaver 1949) is the most common diversity index used in 301(h) programs. It measures the relative distribution of individual organisms among the species present in the sample. *H′* increases for broader distributions of individuals among species. If all individuals are of one species then *H′* is 0.00. If each individual organism is a separate species then *H′* is determined by the logarithm of the total number of organisms collected. For other distributions, the diversity index is given by: $H' = -\sum_{j=1}^{S}\left[\left(\frac{n_j}{N}\right)\ln\left(\frac{n_j}{N}\right)\right]$; where: S = total number of species, n_j = number of individuals in the jth species, N = total number of individuals, and ln = natural logarithm (base e). The value of the Shannon index usually falls between 1.5 and 3.5 and only rarely surpasses 4.5 (Margalef 1972).

Some studies have found a decrease in this index in response to pollutant stress in benthic communities. However, this index is ambiguous because it depends on how evenly the organisms are distributed among the species. Consequently, a statistically significant reduction in this index can occur in the absence of anthropogenic (human-induced) stresses.

When randomness in the sample cannot be guaranteed, then the Brillouin index (*h*) is a more appropriate measure of diversity (Pielou 1977). It is calculated using the formula: $h = \frac{\ln(N!) - \sum_{j=1}^{S}\left[\ln(n_j!)\right]}{N}$. It gives a similar measure of diversity although it produces a slightly lower value than the Shannon index when applied to the same data. In contrast to the Shannon index, the Brillouin will vary between samples although the number of species and their proportional abundance remain the same. Because of its dependence on sample size and the increased complexity of its computation, the Brillouin index is more rarely used than the Shannon index.

1.1.1 Evenness

The Pielou evenness index (*J′*) (Pielou 1977) measures how evenly the individual organisms are distributed among the species present in the sample. *J′* increases for more even distributions of individuals among species. If all individuals belong to a single species, then *J′* is indeterminate. If each species is represented by a single organism, then *J′* will be equal to 1.00. For other distributions, the evenness index is given by: $J' = \frac{H'}{\ln S}$; where: S = total number of species, and H' = Shannon-Wiener diversity index. Because this index is derived from the Shannon-Wiener diversity index (*H′*), it is subject to the same limitations.

1.1.2 Species Dominance

Dominance indices increase with decreasing diversity and are heavily weighted toward the most common species in a sample. The best known of these measures is Simpson's index (*C′*) (Simpson 1949; Wittaker

1965). It is related to other diversity and evenness indices but increases with increasing proportions of individuals associated with a few species. If all individuals are of one species, then C' is maximum and equal to the maximum dominance of 1.00. If individual organisms are evenly distributed among species (J'=1.00), then C' asymptotically approaches 0.00 with increasing numbers of individuals. The Simpson dominance measure is given by: $C' = \sum_{j=1}^{S} \left(\frac{n_j}{N} \right)^2$ but for finite communities, the unbiased form is: $C' = \sum_{j=1}^{S} \left(\frac{n_j (n_j - 1)}{N(N-1)} \right)$ (Magurran 1988).

An unrelated measure of dominance has been ascribed to Swartz et al. (1985). The Swartz dominance index (*Sw*) is defined as the minimum number of species that account for 75% of all individual organisms collected in a sample. In this dominance measure, species are first ranked by the number of individual organisms before establishing the cumulative percent equal to or exceeding 75%. This is an inverse measure because higher dominance is reflected in a lower number of species accounting for 75% of the individual organisms. Despite this, it is not subject to many of the limitations that plague the other univariate indices. It is a non-parametric measure that does not assume an underlying distribution of individuals among species.

1.1.3 Infaunal Trophic Index

The Infaunal Trophic Index (*ITI)* (Word 1978) compares the abundance of four soft-bottom benthic assemblages distinguished by feeding behavior. Because the sensitivity or tolerance to organic enrichment in particulate matter differs among the four groups, shifts in group dominance, as reflected in a changing infaunal index, can be indicative of changed or degraded environmental conditions. The *ITI* ranges between 0 and 100. When species in Group I (suspension feeders) and Group II (surface-detritus feeders) dominate, index values are above 58 and sediments are relatively clean. Lower infaunal indices occur when species in Group III (surface deposit feeders) and Group IV (sub-surface detritus feeders) dominate and sediments are high in organics. The *ITI* is computed from: $ITI = 100 - 33.3 \left[\frac{G_2 + 2G_3 + 3G_4}{G_1 + G_2 + G_3 + G_4} \right]$; where: G_i = number of individual organisms within the i^{th} trophic group.

1.1.4 Species Richness

The Margalef species richness index (d) (Margalef 1951) measures the number of species in a sample relative to the number of individual organisms. d strongly increases for increasing number of species and increases only logarithmically for decreasing number of individuals. If only one species is present then d is 0.00. For other distributions, the richness index is given by: $d = \frac{S - 1}{\ln N}$. Its applicability to biological communities is dependent on whether specimens are log-normally distributed among the species in a given sample. Such an assumption is not globally applicable to benthic marine communities, and without testing each data set, the richness index is of questionable value.

Table E.1 Species Abundance Totaled among Replicate Samples Collected at Benthic Stations in October 2005

Taxonomic Group	Taxon	Station 1	2	3	4	5	6	7	8	9	Station Total	Percent of Total	Cumu-lative %
Crustacea	*Eohaustorius sencillus*	73	58	45	91	52	103	81	68	75	646	13.65	13.65
Crustacea	*Photis macinerneyi*	48	51	37	34	41	91	60	70	148	580	12.25	25.90
Crustacea	*Rhepoxynius abronius*	94	38	38	44	64	48	56	85	22	489	10.33	36.23
Nemertea	*Carinoma mutabilis*	68	34	43	22	32	50	59	56	44	408	8.62	44.86
Crustacea	*Pacifoculodes spinipes*	12	21	25	22	35	54	26	24	16	235	4.97	49.82
Crustacea	*Rhepoxynius menziesi*	7	22	16	24	23	20	26	7	53	198	4.18	54.00
Echinodermata	*Dendraster excentricus*	25	17	14	10	14	23	15	31	19	168	3.55	57.55
Crustacea	*Majoxiphalus major*	4	16	16	31	22	19	13	5	37	163	3.44	61.00
Bivalvia	*Tellina modesta*	23	15	23	12	10	23	24	21	12	163	3.44	64.44
Annelida	*Nephtys caecoides*	14	10	16	12	29	9	17	18	16	141	2.98	67.42
Crustacea	*Tecticeps convexus*	7	14	29	4	8	36	7	11	18	134	2.83	70.25
Annelida	*Scoloplos armiger*	13	10	14	11	6	15	22	10	19	120	2.54	72.79
Annelida	*Glycinde armigera*	18	6	8	7	12	17	8	28	9	113	2.39	75.17
Crustacea	*Mesolamprops dillonensis*	14	12	3	12	27	13	6	9	4	100	2.11	77.29
Bivalvia	*Tellina bodegensis*	20	1	2	6	14	9	6	7	21	86	1.82	79.10
Annelida	*Glycera convoluta*	8	2	8	6	9	16	9	13	9	80	1.69	80.79
Bivalvia	*Nuculana*	4	4	3	8	9	14	9	4	6	61	1.29	82.08
Annelida	*Ampharete labrops*	24	7	1	7	1	6	1	4	4	55	1.16	83.25
Crustacea	*Foxiphalus xiximeus*	16	4	2	7	1	4	2	14	–	50	1.06	84.30
Crustacea	*Americhilidium shoemakeri*	1	3	7	9	5	6	1	5	8	45	0.95	85.25
Annelida	*Lumbrineris californiensis*	4	5	6	6	8	3	5	5	2	44	0.93	86.18
Crustacea	*Photis brevipes*	7	1	1	4	4	1	2	18	1	39	0.82	87.01
Crustacea	*Aoroides inermis*	–	–	–	1	–	38	–	–	–	39	0.82	87.83
Echinodermata	*Amphiodia*	6	2	4	1	–	9	–	4	11	37	0.78	88.61
Bivalvia	*Siliqua lucida*	3	3	3	4	2	3	4	7	7	36	0.76	89.37
Annelida	*Sigalion spinosa*	7	3	2	4	3	5	2	6	1	33	0.70	90.07
Annelida	*Chaetozone setosa*	7	3	–	1	5	2	4	1	7	30	0.63	90.70
Crustacea	*Anchicolurus occidentalis*	6	1	–	6	4	3	2	3	2	27	0.57	91.27
Annelida	*Onuphis iridescens*	7	1	4	–	–	2	2	9	2	27	0.57	91.84
Bivalvia	*Cooperella subdiaphana*	–	1	5	4	2	2	5	4	4	27	0.57	92.41
Annelida	*Spiophanes missionensis*	5	2	2	2	2	4	3	2	–	22	0.46	92.88
Annelida	*Apoprionospio pygmaea*	2	–	–	–	4	6	1	4	3	20	0.42	93.30
Nemertea	*Cerebratulus californiensis*	–	–	–	1	5	6	1	3	2	18	0.38	93.68
Crustacea	*Gibberosus myersi*	3	1	–	4	1	–	1	4	1	15	0.32	94.00
Annelida	*Chone mollis*	2	1	1	2	1	–	4	1	1	13	0.27	94.27
Annelida	*Magelona sacculata*	2	1	–	1	3	2	1	2	1	13	0.27	94.55
Bivalvia	*Chione*	–	–	–	–	1	4	3	4	–	12	0.25	94.80
Crustacea	*Edotia sublittoralis*	1	–	–	–	4	1	–	3	1	10	0.21	95.01
Annelida	*Spiophanes bombyx*	2	1	–	1	3	1	–	2	–	10	0.21	95.23
Crustacea	*Tiron biocellata*	–	–	–	1	2	–	–	7	–	10	0.21	95.44
Crustacea	Isaeidae	9	–	–	–	–	–	–	–	–	9	0.19	95.63
Crustacea	*Synidotea bicuspida*	–	1	–	4	–	1	1	1	1	9	0.19	95.82
Vertebrata	*Porichthys notatus*	–	1	–	2	5	–	1	–	–	9	0.19	96.01
Annelida	*Eumida*	1	–	–	–	–	3	1	3	–	8	0.17	96.18
Mollusca	*Cadulus tolmiei*	8	–	–	–	–	–	–	–	–	8	0.17	96.34

Table E.1 Species Abundance Totaled among Replicate Samples Collected at Benthic Stations in October 2005

Taxonomic Group	Taxon	Station 1	2	3	4	5	6	7	8	9	Station Total	Percent of Total	Cumu-lative %
Crustacea	*Leuroleberis sharpei*	–	–	–	2	1	2	1	2	–	8	0.17	96.51
Mollusca	*Turbonilla*	–	–	–	–	–	3	2	1	2	8	0.17	96.68
Crustacea	*Cyclaspis nubila*	1	–	–	1	1	2	–	2	–	7	0.15	96.83
Phoronida	*Phoronis*	–	–	–	–	1	1	1	4	–	7	0.15	96.98
Crustacea	*Bathycopea daltonae*	2	–	–	3	–	1	–	–	–	6	0.13	97.11
Nemertea	*Paranemertes sp. A*	3	–	–	–	–	–	–	3	–	6	0.13	97.23
Annelida	*Phyllodoce*	–	–	–	–	–	2	1	3	–	6	0.13	97.36
Crustacea	*Diastylopsis tenuis*	–	1	–	1	1	2	–	–	–	5	0.11	97.46
Mollusca	*Nassarius fossatus*	–	–	1	–	1	1	1	1	–	5	0.11	97.57
Bivalvia	*Rochefortia tumida*	–	–	–	–	2	1	–	–	2	5	0.11	97.68
Nemertea	*Tubulanus*	2	–	–	–	–	1	–	1	–	4	0.08	97.76
Cnidaria	Edwardsiidae	4	–	–	–	–	–	–	–	–	4	0.08	97.84
Crustacea	Lysianassidae	–	–	–	1	–	1	–	2	–	4	0.08	97.93
Bivalvia	*Macoma acolasta*	2	–	–	–	–	1	–	–	–	3	0.06	97.99
Bivalvia	*Solen sicarius*	2	–	–	–	–	–	–	1	–	3	0.06	98.06
Crustacea	*Cheirimedia zotea*	3	–	–	–	–	–	–	–	–	3	0.06	98.12
Nemertea	Anopla	3	–	–	–	–	–	–	–	–	3	0.06	98.18
Annelida	*Axiothella rubrocincta*	–	1	–	–	–	1	–	1	–	3	0.06	98.25
Crustacea	Oedicerotidae	–	2	–	1	–	–	–	–	–	3	0.06	98.31
Echinodermata	*Amphiodia digitata*	–	–	–	1	–	–	–	2	–	3	0.06	98.37
Crustacea	*Crangon alaskensis*	–	–	–	–	1	1	–	1	–	3	0.06	98.44
Crustacea	Amphipoda	–	–	–	–	2	1	–	–	–	3	0.06	98.50
Crustacea	*Isocheles pilosus*	–	–	–	–	–	–	–	3	–	3	0.06	98.56
Bivalvia	*Pandora punctata*	1	1	–	–	–	–	–	–	–	2	0.04	98.61
Annelida	*Polydora*	2	–	–	–	–	–	–	–	–	2	0.04	98.65
Cnidaria	Actiniaria	2	–	–	–	–	–	–	–	–	2	0.04	98.69
Annelida	*Armandia bioculata*	2	–	–	–	–	–	–	–	–	2	0.04	98.73
Mollusca	*Balcis rutila*	2	–	–	–	–	–	–	–	–	2	0.04	98.77
Bivalvia	Bivalvia	2	–	–	–	–	–	–	–	–	2	0.04	98.82
Crustacea	*Neomysis kadiakensis*	–	2	–	–	–	–	–	–	–	2	0.04	98.86
Crustacea	*Cancer*	–	–	1	–	–	–	–	1	–	2	0.04	98.90
Crustacea	*Bathyleberis californica*	–	–	–	1	–	–	1	–	–	2	0.04	98.94
Bivalvia	*Pandora bilirata*	–	–	–	1	–	–	–	1	–	2	0.04	98.99
Mollusca	*Odostomia*	–	–	–	–	–	2	–	–	–	2	0.04	99.03
Crustacea	*Ericthonius brasiliensis*	–	–	–	–	–	2	–	–	–	2	0.04	99.07
Echinodermata	*Leptosynapta*	–	–	–	–	–	–	2	–	–	2	0.04	99.11
Mollusca	*Olivella pedroana*	–	–	–	–	–	–	2	–	–	2	0.04	99.15
Mollusca	*Polinices lewisii*	–	–	–	–	–	–	–	1	1	2	0.04	99.20
Nemertea	*Amphiporus bimaculatus*	–	–	–	–	–	–	–	2	–	2	0.04	99.24
Nemertea	Hoplonemertea	–	–	–	–	–	–	–	–	2	2	0.04	99.28
Annelida	*Typosyllis farallonensis*	1	–	–	–	–	–	–	–	–	1	0.02	99.30
Annelida	*Goniada brunnea*	1	–	–	–	–	–	–	–	–	1	0.02	99.32
Annelida	*Podarkeopsis glabra*	1	–	–	–	–	–	–	–	–	1	0.02	99.35
Annelida	*Platynereis bicanaliculata*	1	–	–	–	–	–	–	–	–	1	0.02	99.37
Annelida	Capitellidae	1	–	–	–	–	–	–	–	–	1	0.02	99.39

Table E.1 Species Abundance Totaled among Replicate Samples Collected at Benthic Stations in October 2005

Taxonomic Group	Taxon	Station 1	2	3	4	5	6	7	8	9	Station Total	Percent of Total	Cumu-lative %
Annelida	*Malmgreniella*	1	–	–	–	–	–	–	–	–	1	0.02	99.41
Annelida	*Owenia collaris*	1	–	–	–	–	–	–	–	–	1	0.02	99.43
Echinodermata	Holothuroidea	1	–	–	–	–	–	–	–	–	1	0.02	99.45
Annelida	*Lumbrineris*	1	–	–	–	–	–	–	–	–	1	0.02	99.47
Annelida	*Pectinaria californiensis*	–	1	–	–	–	–	–	–	–	1	0.02	99.49
Annelida	Nereidae	–	1	–	–	–	–	–	–	–	1	0.02	99.51
Crustacea	*Photis*	–	–	–	1	–	–	–	–	–	1	0.02	99.54
Crustacea	*Mandibulophoxus gilesi*	–	–	–	–	1	–	–	–	–	1	0.02	99.56
Crustacea	*Synidotea*	–	–	–	–	1	–	–	–	–	1	0.02	99.58
Crustacea	Cirolanidae	–	–	–	–	1	–	–	–	–	1	0.02	99.60
Annelida	Maldanidae	–	–	–	–	–	1	–	–	–	1	0.02	99.62
Annelida	*Sthenelais verruculosa*	–	–	–	–	–	1	–	–	–	1	0.02	99.64
Crustacea	Gammaridea	–	–	–	–	–	1	–	–	–	1	0.02	99.66
Crustacea	*Blepharipoda occidentalis*	–	–	–	–	–	1	–	–	–	1	0.02	99.68
Annelida	*Scalibregma californicum*	–	–	–	–	–	–	1	–	–	1	0.02	99.70
Bivalvia	*Clinocardium nuttallii*	–	–	–	–	–	–	1	–	–	1	0.02	99.73
Annelida	*Eteone*	–	–	–	–	–	–	1	–	–	1	0.02	99.75
Crustacea	*Listriella melanica*	–	–	–	–	–	–	1	–	–	1	0.02	99.77
Annelida	*Nephtys cornuta*	–	–	–	–	–	–	–	1	–	1	0.02	99.79
Annelida	*Tenonia priops*	–	–	–	–	–	–	–	1	–	1	0.02	99.81
Mollusca	Columbellidae	–	–	–	–	–	–	–	1	–	1	0.02	99.83
Annelida	*Spiochaetopterus costarum*	–	–	–	–	–	–	–	1	–	1	0.02	99.85
Annelida	*Syllides*	–	–	–	–	–	–	–	1	–	1	0.02	99.87
Crustacea	Cumacea	–	–	–	–	–	–	–	1	–	1	0.02	99.89
Mollusca	Gastropoda	–	–	–	–	–	–	–	1	–	1	0.02	99.92
Bivalvia	Mytilidae	–	–	–	–	–	–	–	1	–	1	0.02	99.94
Annelida	*Diopatra ornata*	–	–	–	–	–	–	–	–	1	1	0.02	99.96
Nemertea	*Micrura alaskensis*	–	–	–	–	–	–	–	–	1	1	0.02	99.98
Mollusca	*Rictaxis punctocaelatus*	–	–	–	–	–	–	–	–	1	1	0.02	100.00
Total Number of Individuals		**617**	**382**	**380**	**441**	**486**	**701**	**506**	**623**	**597**	**4733**		

Table E.2 Species Abundance within Replicate Samples Collected at Benthic Station 1 in October 2005

Taxonomic Group	Taxon	Replicate 1	2	3	4	5	Total	Percent of Total	Cumulative Percent
Crustacea	*Rhepoxynius abronius*	16	28	29	12	9	94	15.24	15.24
Crustacea	*Eohaustorius sencillus*	5	22	9	18	19	73	11.83	27.07
Nemertea	*Carinoma mutabilis*	9	17	13	15	14	68	11.02	38.09
Crustacea	*Photis macinerneyi*	6	12	4	25	1	48	7.78	45.87
Echinodermata	*Dendraster excentricus*	3	8	7	4	3	25	4.05	49.92
Annelida	*Ampharete labrops*	12	2	–	6	4	24	3.89	53.81
Bivalvia	*Tellina modesta*	4	4	9	3	3	23	3.73	57.54
Bivalvia	*Tellina bodegensis*	3	7	6	3	1	20	3.24	60.78
Annelida	*Glycinde armigera*	1	5	5	5	2	18	2.92	63.70
Crustacea	*Foxiphalus xiximeus*	10	2	2	2	–	16	2.59	66.29
Annelida	*Nephtys caecoides*	2	2	4	5	1	14	2.27	68.56
Crustacea	*Mesolamprops dillonensis*	1	7	1	3	2	14	2.27	70.83
Annelida	*Scoloplos armiger*	2	4	3	4	–	13	2.11	72.93
Crustacea	*Pacifoculodes spinipes*	1	7	2	1	1	12	1.94	74.88
Crustacea	Isaeidae	8	–	–	1	–	9	1.46	76.34
Annelida	*Glycera convoluta*	2	2	1	3	–	8	1.30	77.63
Mollusca	*Cadulus tolmiei*	1	–	–	4	3	8	1.30	78.93
Annelida	*Onuphis iridescens*	4	–	1	–	2	7	1.13	80.06
Annelida	*Sigalion spinosa*	3	2	–	2	–	7	1.13	81.20
Crustacea	*Photis brevipes*	3	–	2	2	–	7	1.13	82.33
Crustacea	*Tecticeps convexus*	–	3	2	–	2	7	1.13	83.47
Annelida	*Chaetozone setosa*	–	3	1	3	–	7	1.13	84.60
Crustacea	*Rhepoxynius menziesi*	–	2	3	1	1	7	1.13	85.74
Echinodermata	*Amphiodia*	2	–	–	3	1	6	0.97	86.71
Crustacea	*Anchicolurus occidentalis*	–	5	–	1	–	6	0.97	87.68
Annelida	*Spiophanes missionensis*	–	1	1	1	2	5	0.81	88.49
Annelida	*Lumbrineris californiensis*	2	–	1	1	–	4	0.65	89.14
Cnidaria	Edwardsiidae	1	2	–	1	–	4	0.65	89.79
Bivalvia	*Nuculana*	–	3	1	–	–	4	0.65	90.44
Crustacea	*Majoxiphalus major*	–	–	2	1	1	4	0.65	91.09
Crustacea	*Cheirimedia zotea*	3	–	–	–	–	3	0.49	91.57
Nemertea	Anopla	3	–	–	–	–	3	0.49	92.06
Nemertea	*Paranemertes sp. A*	1	–	1	1	–	3	0.49	92.54
Bivalvia	*Siliqua lucida*	–	–	2	1	–	3	0.49	93.03
Crustacea	*Gibberosus myersi*	–	–	1	2	–	3	0.49	93.52
Annelida	*Polydora*	2	–	–	–	–	2	0.32	93.84
Cnidaria	Actiniaria	2	–	–	–	–	2	0.32	94.17
Mollusca	*Balcis rutila*	1	–	1	–	–	2	0.32	94.49
Bivalvia	*Macoma acolasta*	–	1	1	–	–	2	0.32	94.81
Nemertea	*Tubulanus*	–	1	1	–	–	2	0.32	95.14

Table E.2 Species Abundance within Replicate Samples Collected at Benthic Station 1 in October 2005

Taxonomic Group	Taxon	Replicate 1	2	3	4	5	Total	Percent of Total	Cumulative Percent
Bivalvia	*Solen sicarius*	–	1	–	1	–	2	0.32	95.46
Bivalvia	Bivalvia	–	1	–	1	–	2	0.32	95.79
Annelida	*Magelona sacculata*	–	1	–	1	–	2	0.32	96.11
Annelida	*Chone mollis*	–	1	–	–	1	2	0.32	96.43
Annelida	*Spiophanes bombyx*	–	–	1	1	–	2	0.32	96.76
Crustacea	*Bathycopea daltonae*	–	–	1	1	–	2	0.32	97.08
Annelida	*Apoprionospio pygmaea*	–	–	1	–	1	2	0.32	97.41
Annelida	*Armandia bioculata*	–	–	–	2	–	2	0.32	97.73
Annelida	*Podarkeopsis glabra*	1	–	–	–	–	1	0.16	97.89
Crustacea	*Edotia sublittoralis*	1	–	–	–	–	1	0.16	98.06
Annelida	*Eumida*	1	–	–	–	–	1	0.16	98.22
Annelida	*Owenia collaris*	1	–	–	–	–	1	0.16	98.38
Annelida	*Typosyllis farallonensis*	–	1	–	–	–	1	0.16	98.54
Annelida	*Goniada brunnea*	–	–	1	–	–	1	0.16	98.70
Bivalvia	*Pandora punctata*	–	–	1	–	–	1	0.16	98.87
Annelida	Capitellidae	–	–	1	–	–	1	0.16	99.03
Crustacea	*Americhilidium shoemakeri*	–	–	–	1	–	1	0.16	99.19
Crustacea	*Cyclaspis nubila*	–	–	–	1	–	1	0.16	99.35
Annelida	*Platynereis bicanaliculata*	–	–	–	1	–	1	0.16	99.51
Annelida	*Malmgreniella*	–	–	–	1	–	1	0.16	99.68
Annelida	*Lumbrineris*	–	–	–	1	–	1	0.16	99.84
Echinodermata	Holothuroidea	–	–	–	–	1	1	0.16	100.00
Total Number of Individuals		**117**	**157**	**122**	**146**	**75**	**617**		

Table E.3 Species Abundance within Replicate Samples Collected at Benthic Station 2 in October 2005

Taxonomic Group	Taxon	Replicate 1	2	3	4	5	Total	Percent of Total	Cumulative Percent
Crustacea	*Eohaustorius sencillus*	8	22	17	9	2	58	15.18	15.18
Crustacea	*Photis macinerneyi*	27	2	14	4	4	51	13.35	28.53
Crustacea	*Rhepoxynius abronius*	4	15	8	9	2	38	9.95	38.48
Nemertea	*Carinoma mutabilis*	8	7	6	4	9	34	8.90	47.38
Crustacea	*Rhepoxynius menziesi*	6	3	2	6	5	22	5.76	53.14
Crustacea	*Pacifoculodes spinipes*	6	–	8	6	1	21	5.50	58.64
Echinodermata	*Dendraster excentricus*	3	2	7	2	3	17	4.45	63.09

Table E.3 Species Abundance within Replicate Samples Collected at Benthic Station 2 in October 2005

Taxonomic Group	Taxon	Replicate 1	2	3	4	5	Total	Percent of Total	Cumulative Percent
Crustacea	*Majoxiphalus major*	5	6	1	3	1	16	4.19	67.28
Bivalvia	*Tellina modesta*	3	3	5	3	1	15	3.93	71.20
Crustacea	*Tecticeps convexus*	8	–	1	3	2	14	3.66	74.87
Crustacea	*Mesolamprops dillonensis*	2	1	5	3	1	12	3.14	78.01
Annelida	*Nephtys caecoides*	4	1	3	–	2	10	2.62	80.63
Annelida	*Scoloplos armiger*	1	3	2	1	3	10	2.62	83.25
Annelida	*Ampharete labrops*	6	–	1	–	–	7	1.83	85.08
Annelida	*Glycinde armigera*	2	2	–	1	1	6	1.57	86.65
Annelida	*Lumbrineris californiensis*	–	2	3	–	–	5	1.31	87.96
Crustacea	*Foxiphalus xiximeus*	2	1	1	–	–	4	1.05	89.01
Bivalvia	*Nuculana*	1	2	–	–	1	4	1.05	90.05
Bivalvia	*Siliqua lucida*	1	2	–	–	–	3	0.79	90.84
Annelida	*Chaetozone setosa*	1	1	–	1	–	3	0.79	91.62
Annelida	*Sigalion spinosa*	–	–	3	–	–	3	0.79	92.41
Crustacea	*Americhilidium shoemakeri*	–	–	1	1	1	3	0.79	93.19
Annelida	*Spiophanes missionensis*	2	–	–	–	–	2	0.52	93.72
Echinodermata	*Amphiodia*	1	1	–	–	–	2	0.52	94.24
Annelida	*Glycera convoluta*	1	–	–	1	–	2	0.52	94.76
Crustacea	Oedicerotidae	1	–	–	–	1	2	0.52	95.29
Crustacea	*Neomysis kadiakensis*	–	1	–	–	1	2	0.52	95.81
Annelida	*Pectinaria californiensis*	1	–	–	–	–	1	0.26	96.07
Vertebrata	*Porichthys notatus*	1	–	–	–	–	1	0.26	96.34
Crustacea	*Gibberosus myersi*	1	–	–	–	–	1	0.26	96.60
Crustacea	*Synidotea bicuspida*	1	–	–	–	–	1	0.26	96.86
Annelida	*Magelona sacculata*	1	–	–	–	–	1	0.26	97.12
Annelida	*Onuphis iridescens*	1	–	–	–	–	1	0.26	97.38
Crustacea	*Photis brevipes*	–	1	–	–	–	1	0.26	97.64
Bivalvia	*Tellina bodegensis*	–	1	–	–	–	1	0.26	97.91
Annelida	*Axiothella rubrocincta*	–	1	–	–	–	1	0.26	98.17
Bivalvia	*Cooperella subdiaphana*	–	1	–	–	–	1	0.26	98.43
Annelida	*Spiophanes bombyx*	–	–	1	–	–	1	0.26	98.69
Bivalvia	*Pandora punctata*	–	–	1	–	–	1	0.26	98.95
Crustacea	*Anchicolurus occidentalis*	–	–	1	–	–	1	0.26	99.21
Crustacea	*Diastylopsis tenuis*	–	–	1	–	–	1	0.26	99.48
Annelida	Nereidae	–	–	1	–	–	1	0.26	99.74
Annelida	*Chone mollis*	–	–	–	–	1	1	0.26	100.00
Total Number of Individuals		**109**	**81**	**93**	**57**	**42**	**382**		

Table E.4 Species Abundance within Replicate Samples Collected at Benthic Station 3 in October 2005

Taxonomic Group	Taxon	Replicate 1	Replicate 2	Replicate 3	Replicate 4	Replicate 5	Total	Percent of Total	Cumulative Percent
Crustacea	*Eohaustorius sencillus*	11	12	16	–	6	45	11.84	11.84
Nemertea	*Carinoma mutabilis*	8	13	4	10	8	43	11.32	23.16
Crustacea	*Rhepoxynius abronius*	9	2	10	11	6	38	10.00	33.16
Crustacea	*Photis macinerneyi*	2	17	–	13	5	37	9.74	42.89
Crustacea	*Tecticeps convexus*	3	3	1	2	20	29	7.63	50.53
Crustacea	*Pacifoculodes spinipes*	3	6	1	5	10	25	6.58	57.11
Bivalvia	*Tellina modesta*	7	2	3	4	7	23	6.05	63.16
Crustacea	*Rhepoxynius menziesi*	4	2	2	3	5	16	4.21	67.37
Crustacea	*Majoxiphalus major*	2	3	3	3	5	16	4.21	71.58
Annelida	*Nephtys caecoides*	2	1	8	5	–	16	4.21	75.79
Annelida	*Scoloplos armiger*	4	2	3	3	2	14	3.68	79.47
Echinodermata	*Dendraster excentricus*	–	7	2	3	2	14	3.68	83.16
Annelida	*Glycinde armigera*	4	1	1	2	–	8	2.11	85.26
Annelida	*Glycera convoluta*	–	2	1	1	4	8	2.11	87.37
Crustacea	*Americhilidium shoemakeri*	–	5	1	1	–	7	1.84	89.21
Annelida	*Lumbrineris californiensis*	3	3	–	–	–	6	1.58	90.79
Bivalvia	*Cooperella subdiaphana*	1	2	1	–	1	5	1.32	92.11
Annelida	*Onuphis iridescens*	1	1	1	1	–	4	1.05	93.16
Echinodermata	*Amphiodia*	1	–	–	3	–	4	1.05	94.21
Bivalvia	*Siliqua lucida*	1	1	–	1	–	3	0.79	95.00
Bivalvia	*Nuculana*	1	–	1	–	1	3	0.79	95.79
Crustacea	*Mesolamprops dillonensis*	–	2	–	–	1	3	0.79	96.58
Crustacea	*Foxiphalus xiximeus*	–	1	–	–	1	2	0.53	97.11
Bivalvia	*Tellina bodegensis*	–	–	1	–	1	2	0.53	97.63
Annelida	*Sigalion spinosa*	–	–	–	1	1	2	0.53	98.16
Annelida	*Spiophanes missionensis*	–	–	–	–	2	2	0.53	98.68
Annelida	*Ampharete labrops*	1	–	–	–	–	1	0.26	98.95
Mollusca	*Nassarius fossatus*	1	–	–	–	–	1	0.26	99.21
Annelida	*Chone mollis*	–	–	1	–	–	1	0.26	99.47
Crustacea	*Cancer*	–	–	–	1	–	1	0.26	99.74
Crustacea	*Photis brevipes*	–	–	–	–	1	1	0.26	100.00
Total Number of Individuals		**69**	**88**	**61**	**73**	**89**	**380**		

Table E.5 Species Abundance within Replicate Samples Collected at Benthic Station 4 in October 2005

Taxonomic		Replicate						Percent	Cumulative
Group	Taxon	1	2	3	4	5	Total	of Total	Percent
Crustacea	*Eohaustorius sencillus*	14	19	14	8	36	91	20.63	20.63
Crustacea	*Rhepoxynius abronius*	7	7	10	9	11	44	9.98	30.61
Crustacea	*Photis macinerneyi*	7	7	4	12	4	34	7.71	38.32
Crustacea	*Majoxiphalus major*	12	5	8	4	2	31	7.03	45.35
Crustacea	*Rhepoxynius menziesi*	1	3	8	11	1	24	5.44	50.79
Crustacea	*Pacifoculodes spinipes*	4	9	1	2	6	22	4.99	55.78
Nemertea	*Carinoma mutabilis*	3	3	3	9	4	22	4.99	60.77
Bivalvia	*Tellina modesta*	2	1	3	4	2	12	2.72	63.49
Annelida	*Nephtys caecoides*	1	4	4	2	1	12	2.72	66.21
Crustacea	*Mesolamprops dillonensis*	1	2	3	4	2	12	2.72	68.93
Annelida	*Scoloplos armiger*	2	1	2	3	3	11	2.49	71.43
Echinodermata	*Dendraster excentricus*	5	–	–	2	3	10	2.27	73.70
Crustacea	*Americhilidium shoemakeri*	4	1	1	3	–	9	2.04	75.74
Bivalvia	*Nuculana*	3	1	1	2	1	8	1.81	77.55
Annelida	*Glycinde armigera*	3	3	–	–	1	7	1.59	79.14
Crustacea	*Foxiphalus xiximeus*	1	–	2	4	–	7	1.59	80.73
Annelida	*Ampharete labrops*	–	3	1	3	–	7	1.59	82.31
Bivalvia	*Tellina bodegensis*	2	–	1	2	1	6	1.36	83.67
Annelida	*Lumbrineris californiensis*	2	–	1	–	3	6	1.36	85.03
Crustacea	*Anchicolurus occidentalis*	1	–	2	3	–	6	1.36	86.39
Annelida	*Glycera convoluta*	–	–	1	3	2	6	1.36	87.76
Bivalvia	*Cooperella subdiaphana*	2	–	–	1	1	4	0.91	88.66
Crustacea	*Tecticeps convexus*	1	1	2	–	–	4	0.91	89.57
Annelida	*Sigalion spinosa*	1	–	–	2	1	4	0.91	90.48
Bivalvia	*Siliqua lucida*	1	–	1	1	1	4	0.91	91.38
Crustacea	*Synidotea bicuspida*	–	3	–	1	–	4	0.91	92.29
Crustacea	*Photis brevipes*	–	2	1	–	1	4	0.91	93.20
Crustacea	*Gibberosus myersi*	–	–	–	4	–	4	0.91	94.10
Crustacea	*Bathycopea daltonae*	1	–	1	1	–	3	0.68	94.78
Annelida	*Spiophanes missionensis*	1	–	–	1	–	2	0.45	95.24
Crustacea	*Leuroleberis sharpei*	1	–	–	1	–	2	0.45	95.69
Vertebrata	*Porichthys notatus*	–	–	2	–	–	2	0.45	96.15
Annelida	*Chone mollis*	–	–	–	1	1	2	0.45	96.60
Annelida	*Spiophanes bombyx*	1	–	–	–	–	1	0.23	96.83
Echinodermata	*Amphiodia*	1	–	–	–	–	1	0.23	97.05
Annelida	*Chaetozone setosa*	1	–	–	–	–	1	0.23	97.28
Crustacea	*Tiron biocellata*	–	1	–	–	–	1	0.23	97.51
Crustacea	*Bathyleberis californica*	–	1	–	–	–	1	0.23	97.73
Nemertea	*Cerebratulus californiensis*	–	1	–	–	–	1	0.23	97.96
Annelida	*Magelona sacculata*	–	1	–	–	–	1	0.23	98.19

Table E.5 Species Abundance within Replicate Samples Collected at Benthic Station 4 in October 2005

Taxonomic Group	Taxon	Replicate 1	2	3	4	5	Total	Percent of Total	Cumulative Percent
Echinodermata	*Amphiodia digitata*	–	–	–	1	–	1	0.23	98.41
Crustacea	*Cyclaspis nubila*	–	–	–	1	–	1	0.23	98.64
Crustacea	*Aoroides inermis*	–	–	–	1	–	1	0.23	98.87
Crustacea	*Diastylopsis tenuis*	–	–	–	1	–	1	0.23	99.09
Crustacea	Lysianassidae	–	–	–	1	–	1	0.23	99.32
Crustacea	Oedicerotidae	–	–	–	1	–	1	0.23	99.55
Bivalvia	*Pandora bilirata*	–	–	–	–	1	1	0.23	99.77
Crustacea	*Photis*	–	–	–	–	1	1	0.23	100.00
Total Number of Individuals		**86**	**79**	**77**	**109**	**90**	**441**		

Table E.6 Species Abundance within Replicate Samples Collected at Benthic Station 5 in October 2005

Taxonomic Group	Taxon	Replicate 1	2	3	4	5	Total	Percent of Total	Cumulative Percent
Crustacea	*Rhepoxynius abronius*	13	10	20	11	10	64	13.17	13.17
Crustacea	*Eohaustorius sencillus*	24	5	11	6	6	52	10.70	23.87
Crustacea	*Photis macinerneyi*	4	7	23	1	6	41	8.44	32.30
Crustacea	*Pacifoculodes spinipes*	11	9	6	6	3	35	7.20	39.51
Nemertea	*Carinoma mutabilis*	10	1	9	4	8	32	6.58	46.09
Annelida	*Nephtys caecoides*	8	7	4	2	8	29	5.97	52.06
Crustacea	*Mesolamprops dillonensis*	10	2	9	3	3	27	5.56	57.61
Crustacea	*Rhepoxynius menziesi*	12	6	2	1	2	23	4.73	62.35
Crustacea	*Majoxiphalus major*	3	4	4	4	7	22	4.53	66.87
Echinodermata	*Dendraster excentricus*	2	4	2	2	4	14	2.88	69.75
Bivalvia	*Tellina bodegensis*	–	3	7	2	2	14	2.88	72.63
Annelida	*Glycinde armigera*	–	3	4	1	4	12	2.47	75.10
Bivalvia	*Tellina modesta*	3	5	–	1	1	10	2.06	77.16
Bivalvia	*Nuculana*	1	3	2	1	2	9	1.85	79.01
Annelida	*Glycera convoluta*	–	5	–	2	2	9	1.85	80.86
Annelida	*Lumbrineris californiensis*	1	4	1	1	1	8	1.65	82.51
Crustacea	*Tecticeps convexus*	–	3	2	1	2	8	1.65	84.16
Annelida	*Scoloplos armiger*	1	–	1	1	3	6	1.23	85.39
Annelida	*Chaetozone setosa*	3	1	–	1	–	5	1.03	86.42
Nemertea	*Cerebratulus californiensis*	2	–	–	1	2	5	1.03	87.45
Crustacea	*Americhilidium shoemakeri*	–	2	1	1	1	5	1.03	88.48
Vertebrata	*Porichthys notatus*	–	–	–	1	4	5	1.03	89.51

Table E.6 Species Abundance within Replicate Samples Collected at Benthic Station 5 in October 2005

Taxonomic Group	Taxon	Replicate 1	2	3	4	5	Total	Percent of Total	Cumulative Percent
Crustacea	*Edotia sublittoralis*	1	1	–	–	2	4	0.82	90.33
Crustacea	*Anchicolurus occidentalis*	1	1	1	–	1	4	0.82	91.15
Crustacea	*Photis brevipes*	1	–	3	–	–	4	0.82	91.98
Annelida	*Apoprionospio pygmaea*	–	–	3	1	–	4	0.82	92.80
Annelida	*Magelona sacculata*	2	–	1	–	–	3	0.62	93.42
Annelida	*Sigalion spinosa*	1	1	–	–	1	3	0.62	94.03
Annelida	*Spiophanes bombyx*	–	1	–	1	1	3	0.62	94.65
Bivalvia	*Rochefortia tumida*	1	1	–	–	–	2	0.41	95.06
Bivalvia	*Cooperella subdiaphana*	1	–	–	1	–	2	0.41	95.47
Crustacea	*Tiron biocellata*	–	1	1	–	–	2	0.41	95.88
Crustacea	Amphipoda	–	–	2	–	–	2	0.41	96.30
Bivalvia	*Siliqua lucida*	–	–	1	–	1	2	0.41	96.71
Annelida	*Spiophanes missionensis*	–	–	–	2	–	2	0.41	97.12
Crustacea	*Leuroleberis sharpei*	1	–	–	–	–	1	0.21	97.33
Crustacea	*Mandibulophoxus gilesi*	–	1	–	–	–	1	0.21	97.53
Crustacea	*Synidotea*	–	1	–	–	–	1	0.21	97.74
Crustacea	*Foxiphalus xiximeus*	–	1	–	–	–	1	0.21	97.94
Crustacea	*Diastylopsis tenuis*	–	1	–	–	–	1	0.21	98.15
Annelida	*Ampharete labrops*	–	–	1	–	–	1	0.21	98.35
Crustacea	*Cyclaspis nubila*	–	–	1	–	–	1	0.21	98.56
Crustacea	*Crangon alaskensis*	–	–	1	–	–	1	0.21	98.77
Phoronida	*Phoronis*	–	–	–	1	–	1	0.21	98.97
Crustacea	*Gibberosus myersi*	–	–	–	1	–	1	0.21	99.18
Crustacea	Cirolanidae	–	–	–	1	–	1	0.21	99.38
Bivalvia	*Chione*	–	–	–	1	–	1	0.21	99.59
Mollusca	*Nassarius fossatus*	–	–	–	1	–	1	0.21	99.79
Annelida	*Chone mollis*	–	–	–	–	1	1	0.21	100.00
Total Number of Individuals		**117**	**94**	**123**	**64**	**88**	**486**		

Table E.7 Species Abundance within Replicate Samples Collected at Benthic Station 6 in October 2005

Taxonomic Group	Taxon	Replicate 1	2	3	4	5	Total	Percent of Total	Cumulative Percent
Crustacea	*Eohaustorius sencillus*	11	33	27	4	28	103	14.69	14.69
Crustacea	*Photis macinerneyi*	6	32	18	23	12	91	12.98	27.67
Crustacea	*Pacifoculodes spinipes*	7	11	14	18	4	54	7.70	35.38
Nemertea	*Carinoma mutabilis*	14	9	8	10	9	50	7.13	42.51
Crustacea	*Rhepoxynius abronius*	21	11	3	1	12	48	6.85	49.36

Table E.7 Species Abundance within Replicate Samples Collected at Benthic Station 6 in October 2005

Taxonomic Group	Taxon	Replicate 1	2	3	4	5	Total	Percent of Total	Cumulative Percent
Crustacea	*Aoroides inermis*	–	–	–	38	–	38	5.42	54.78
Crustacea	*Tecticeps convexus*	2	10	21	3	–	36	5.14	59.91
Bivalvia	*Tellina modesta*	12	1	6	4	–	23	3.28	63.20
Echinodermata	*Dendraster excentricus*	2	12	4	2	3	23	3.28	66.48
Crustacea	*Rhepoxynius menziesi*	6	3	3	3	5	20	2.85	69.33
Crustacea	*Majoxiphalus major*	1	6	–	4	8	19	2.71	72.04
Annelida	*Glycinde armigera*	1	2	3	6	5	17	2.43	74.47
Annelida	*Glycera convoluta*	5	–	3	5	3	16	2.28	76.75
Annelida	*Scoloplos armiger*	3	5	2	2	3	15	2.14	78.89
Bivalvia	*Nuculana*	1	3	4	2	4	14	2.00	80.88
Crustacea	*Mesolamprops dillonensis*	2	5	3	1	2	13	1.85	82.74
Bivalvia	*Tellina bodegensis*	5	2	2	–	–	9	1.28	84.02
Annelida	*Nephtys caecoides*	1	3	3	1	1	9	1.28	85.31
Echinodermata	*Amphiodia*	–	2	3	4	–	9	1.28	86.59
Nemertea	*Cerebratulus californiensis*	1	2	–	3	–	6	0.86	87.45
Crustacea	*Americhilidium shoemakeri*	1	1	–	4	–	6	0.86	88.30
Annelida	*Apoprionospio pygmaea*	–	3	1	–	2	6	0.86	89.16
Annelida	*Ampharete labrops*	–	–	1	2	3	6	0.86	90.01
Annelida	*Sigalion spinosa*	1	2	–	1	1	5	0.71	90.73
Annelida	*Spiophanes missionensis*	1	1	2	–	–	4	0.57	91.30
Bivalvia	*Chione*	–	3	–	–	1	4	0.57	91.87
Crustacea	*Foxiphalus xiximeus*	–	–	–	1	3	4	0.57	92.44
Bivalvia	*Siliqua lucida*	1	1	–	1	–	3	0.43	92.87
Mollusca	*Turbonilla*	–	2	–	1	–	3	0.43	93.30
Crustacea	*Anchicolurus occidentalis*	–	2	–	–	1	3	0.43	93.72
Annelida	*Lumbrineris californiensis*	–	–	2	1	–	3	0.43	94.15
Annelida	*Eumida*	–	–	–	–	3	3	0.43	94.58
Crustacea	*Cyclaspis nubila*	1	–	–	–	1	2	0.29	94.86
Annelida	*Chaetozone setosa*	1	–	–	–	1	2	0.29	95.15
Annelida	*Onuphis iridescens*	1	–	1	–	–	2	0.29	95.44
Crustacea	*Leuroleberis sharpei*	–	2	–	–	–	2	0.29	95.72
Annelida	*Phyllodoce*	–	1	–	1	–	2	0.29	96.01
Bivalvia	*Cooperella subdiaphana*	–	1	–	–	1	2	0.29	96.29
Crustacea	*Diastylopsis tenuis*	–	1	–	–	1	2	0.29	96.58
Annelida	*Magelona sacculata*	–	–	1	–	1	2	0.29	96.86
Crustacea	*Ericthonius brasiliensis*	–	–	–	2	–	2	0.29	97.15
Mollusca	*Odostomia*	–	–	–	–	2	2	0.29	97.43
Crustacea	*Edotia sublittoralis*	1	–	–	–	–	1	0.14	97.57
Nemertea	*Tubulanus*	1	–	–	–	–	1	0.14	97.72
Crustacea	*Photis brevipes*	–	1	–	–	–	1	0.14	97.86

Table E.7 Species Abundance within Replicate Samples Collected at Benthic Station 6 in October 2005

Taxonomic Group	Taxon	Replicate 1	2	3	4	5	Total	Percent of Total	Cumulative Percent
Crustacea	*Synidotea bicuspida*	–	1	–	–	–	1	0.14	98.00
Crustacea	*Bathycopea daltonae*	–	1	–	–	–	1	0.14	98.15
Mollusca	*Nassarius fossatus*	–	1	–	–	–	1	0.14	98.29
Annelida	*Spiophanes bombyx*	–	–	1	–	–	1	0.14	98.43
Bivalvia	*Macoma acolasta*	–	–	1	–	–	1	0.14	98.57
Annelida	*Axiothella rubrocincta*	–	–	1	–	–	1	0.14	98.72
Phoronida	*Phoronis*	–	–	–	1	–	1	0.14	98.86
Crustacea	Amphipoda	–	–	–	1	–	1	0.14	99.00
Crustacea	*Crangon alaskensis*	–	–	–	1	–	1	0.14	99.14
Crustacea	Gammaridea	–	–	–	1	–	1	0.14	99.29
Crustacea	*Blepharipoda occidentalis*	–	–	–	1	–	1	0.14	99.43
Crustacea	Lysianassidae	–	–	–	1	–	1	0.14	99.57
Annelida	Maldanidae	–	–	–	1	–	1	0.14	99.71
Bivalvia	*Rochefortia tumida*	–	–	–	1	–	1	0.14	99.86
Annelida	*Sthenelais verruculosa*	–	–	–	–	1	1	0.14	100.00
Total Number of Individuals		**110**	**176**	**138**	**156**	**121**	**701**		

Table E.8 Species Abundance within Replicate Samples Collected at Benthic Station 7 in October 2005

Taxonomic Group	Taxon	Replicate 1	2	3	4	5	Total	Percent of Total	Cumulative Percent
Crustacea	*Eohaustorius sencillus*	24	29	8	14	6	81	16.01	16.01
Crustacea	*Photis macinerneyi*	2	17	24	10	7	60	11.86	27.87
Nemertea	*Carinoma mutabilis*	5	18	9	11	16	59	11.66	39.53
Crustacea	*Rhepoxynius abronius*	16	6	12	11	11	56	11.07	50.59
Crustacea	*Rhepoxynius menziesi*	9	6	2	3	6	26	5.14	55.73
Crustacea	*Pacifoculodes spinipes*	6	3	6	7	4	26	5.14	60.87
Bivalvia	*Tellina modesta*	4	6	4	1	9	24	4.74	65.61
Annelida	*Scoloplos armiger*	6	4	2	6	4	22	4.35	69.96
Annelida	*Nephtys caecoides*	4	1	3	3	6	17	3.36	73.32
Echinodermata	*Dendraster excentricus*	3	2	3	2	5	15	2.96	76.28
Crustacea	*Majoxiphalus major*	3	1	–	5	4	13	2.57	78.85
Annelida	*Glycera convoluta*	1	1	1	2	4	9	1.78	80.63
Bivalvia	*Nuculana*	–	3	3	1	2	9	1.78	82.41
Annelida	*Glycinde armigera*	1	3	–	1	3	8	1.58	83.99
Crustacea	*Tecticeps convexus*	3	1	1	2	–	7	1.38	85.38
Bivalvia	*Tellina bodegensis*	2	1	–	3	–	6	1.19	86.56
Crustacea	*Mesolamprops dillonensis*	–	1	1	1	3	6	1.19	87.75

Table E.8 Species Abundance within Replicate Samples Collected at Benthic Station 7 in October 2005

Taxonomic Group	Taxon	Replicate 1	2	3	4	5	Total	Percent of Total	Cumulative Percent
Annelida	*Lumbrineris californiensis*	2	2	1	–	–	5	0.99	88.74
Bivalvia	*Cooperella subdiaphana*	1	–	2	2	–	5	0.99	89.72
Bivalvia	*Siliqua lucida*	2	–	–	2	–	4	0.79	90.51
Annelida	*Chone mollis*	–	–	1	2	1	4	0.79	91.30
Annelida	*Chaetozone setosa*	–	–	1	2	1	4	0.79	92.09
Annelida	*Spiophanes missionensis*	–	1	1	–	1	3	0.59	92.69
Bivalvia	*Chione*	–	1	2	–	–	3	0.59	93.28
Echinodermata	*Leptosynapta*	1	1	–	–	–	2	0.40	93.68
Mollusca	*Turbonilla*	1	–	–	1	–	2	0.40	94.07
Annelida	*Onuphis iridescens*	1	–	–	–	1	2	0.40	94.47
Annelida	*Sigalion spinosa*	–	2	–	–	–	2	0.40	94.86
Mollusca	*Olivella pedroana*	–	2	–	–	–	2	0.40	95.26
Crustacea	*Anchicolurus occidentalis*	–	–	2	–	–	2	0.40	95.65
Crustacea	*Photis brevipes*	–	–	1	–	1	2	0.40	96.05
Crustacea	*Foxiphalus xiximeus*	–	–	–	1	1	2	0.40	96.44
Vertebrata	*Porichthys notatus*	1	–	–	–	–	1	0.20	96.64
Crustacea	*Synidotea bicuspida*	1	–	–	–	–	1	0.20	96.84
Annelida	*Eteone*	1	–	–	–	–	1	0.20	97.04
Crustacea	*Leuroleberis sharpei*	–	1	–	–	–	1	0.20	97.23
Annelida	*Ampharete labrops*	–	–	1	–	–	1	0.20	97.43
Crustacea	*Gibberosus myersi*	–	–	1	–	–	1	0.20	97.63
Annelida	*Eumida*	–	–	1	–	–	1	0.20	97.83
Annelida	*Apoprionospio pygmaea*	–	–	1	–	–	1	0.20	98.02
Annelida	*Scalibregma californicum*	–	–	1	–	–	1	0.20	98.22
Nemertea	*Cerebratulus californiensis*	–	–	1	–	–	1	0.20	98.42
Bivalvia	*Clinocardium nuttallii*	–	–	1	–	–	1	0.20	98.62
Mollusca	*Nassarius fossatus*	–	–	1	–	–	1	0.20	98.81
Phoronida	*Phoronis*	–	–	–	1	–	1	0.20	99.01
Annelida	*Phyllodoce*	–	–	–	1	–	1	0.20	99.21
Crustacea	*Americhilidium shoemakeri*	–	–	–	1	–	1	0.20	99.41
Crustacea	*Bathyleberis californica*	–	–	–	1	–	1	0.20	99.60
Crustacea	*Listriella melanica*	–	–	–	–	1	1	0.20	99.80
Annelida	*Magelona sacculata*	–	–	–	–	1	1	0.20	100.00
Total Number of Individuals		**100**	**113**	**98**	**97**	**98**	**506**		

Table E.9 Species Abundance within Replicate Samples Collected at Benthic Station 8 in October 2005

Taxonomic Group	Taxon	Replicate 1	2	3	4	5	Total	Percent of Total	Cumulative Percent
Crustacea	*Rhepoxynius abronius*	11	19	21	11	23	85	13.64	13.64
Crustacea	*Photis macinerneyi*	9	22	4	26	9	70	11.24	24.88
Crustacea	*Eohaustorius sencillus*	4	24	19	11	10	68	10.91	35.79
Nemertea	*Carinoma mutabilis*	9	8	16	14	9	56	8.99	44.78
Echinodermata	*Dendraster excentricus*	3	2	14	8	4	31	4.98	49.76
Annelida	*Glycinde armigera*	9	8	5	5	1	28	4.49	54.25
Crustacea	*Pacifoculodes spinipes*	5	2	5	3	9	24	3.85	58.11
Bivalvia	*Tellina modesta*	2	5	8	5	1	21	3.37	61.48
Annelida	*Nephtys caecoides*	2	6	3	2	5	18	2.89	64.37
Crustacea	*Photis brevipes*	1	9	1	6	1	18	2.89	67.26
Crustacea	*Foxiphalus xiximeus*	–	7	–	7	–	14	2.25	69.50
Annelida	*Glycera convoluta*	7	1	1	3	1	13	2.09	71.59
Crustacea	*Tecticeps convexus*	–	4	6	–	1	11	1.77	73.35
Annelida	*Scoloplos armiger*	–	5	2	2	1	10	1.61	74.96
Annelida	*Onuphis iridescens*	3	1	2	2	1	9	1.44	76.40
Crustacea	*Mesolamprops dillonensis*	2	2	–	1	4	9	1.44	77.85
Crustacea	*Tiron biocellata*	2	3	1	1	–	7	1.12	78.97
Bivalvia	*Siliqua lucida*	2	–	1	1	3	7	1.12	80.10
Bivalvia	*Tellina bodegensis*	1	1	2	1	2	7	1.12	81.22
Crustacea	*Rhepoxynius menziesi*	–	4	1	1	1	7	1.12	82.34
Annelida	*Sigalion spinosa*	2	1	2	1	–	6	0.96	83.31
Crustacea	*Americhilidium shoemakeri*	–	3	–	1	1	5	0.80	84.11
Annelida	*Lumbrineris californiensis*	–	1	1	1	2	5	0.80	84.91
Crustacea	*Majoxiphalus major*	–	–	1	3	1	5	0.80	85.71
Annelida	*Apoprionospio pygmaea*	3	1	–	–	–	4	0.64	86.36
Annelida	*Ampharete labrops*	2	2	–	–	–	4	0.64	87.00
Bivalvia	*Chione*	2	1	–	–	1	4	0.64	87.64
Bivalvia	*Nuculana*	2	–	1	1	–	4	0.64	88.28
Phoronida	*Phoronis*	1	2	–	1	–	4	0.64	88.92
Echinodermata	*Amphiodia*	1	–	–	3	–	4	0.64	89.57
Crustacea	*Gibberosus myersi*	1	–	–	3	–	4	0.64	90.21
Bivalvia	*Cooperella subdiaphana*	–	–	2	1	1	4	0.64	90.85
Nemertea	*Paranemertes sp. A*	1	–	1	–	1	3	0.48	91.33
Crustacea	*Anchicolurus occidentalis*	1	–	1	1	–	3	0.48	91.81
Crustacea	*Edotia sublittoralis*	–	3	–	–	–	3	0.48	92.30
Crustacea	*Isocheles pilosus*	–	2	1	–	–	3	0.48	92.78
Annelida	*Phyllodoce*	–	1	2	–	–	3	0.48	93.26
Annelida	*Eumida*	–	1	–	2	–	3	0.48	93.74
Nemertea	*Cerebratulus californiensis*	–	–	2	1	–	3	0.48	94.22

Table E.9 Species Abundance within Replicate Samples Collected at Benthic Station 8 in October 2005

Taxonomic Group	Taxon	Replicate 1	2	3	4	5	Total	Percent of Total	Cumulative Percent
Crustacea	Lysianassidae	1	–	–	–	1	2	0.32	94.54
Crustacea	*Cyclaspis nubila*	–	2	–	–	–	2	0.32	94.86
Nemertea	*Amphiporus bimaculatus*	–	2	–	–	–	2	0.32	95.18
Annelida	*Spiophanes missionensis*	–	1	1	–	–	2	0.32	95.51
Annelida	*Magelona sacculata*	–	1	1	–	–	2	0.32	95.83
Annelida	*Spiophanes bombyx*	–	1	–	1	–	2	0.32	96.15
Echinodermata	*Amphiodia digitata*	–	1	–	1	–	2	0.32	96.47
Crustacea	*Leuroleberis sharpei*	–	–	–	2	–	2	0.32	96.79
Crustacea	*Synidotea bicuspida*	1	–	–	–	–	1	0.16	96.95
Crustacea	*Crangon alaskensis*	1	–	–	–	–	1	0.16	97.11
Nemertea	*Tubulanus*	1	–	–	–	–	1	0.16	97.27
Mollusca	*Turbonilla*	1	–	–	–	–	1	0.16	97.43
Bivalvia	Mytilidae	1	–	–	–	–	1	0.16	97.59
Mollusca	Columbellidae	–	1	–	–	–	1	0.16	97.75
Annelida	*Spiochaetopterus costarum*	–	1	–	–	–	1	0.16	97.91
Annelida	*Syllides*	–	1	–	–	–	1	0.16	98.07
Annelida	*Chaetozone setosa*	–	1	–	–	–	1	0.16	98.23
Mollusca	*Nassarius fossatus*	–	1	–	–	–	1	0.16	98.39
Bivalvia	*Solen sicarius*	–	–	1	–	–	1	0.16	98.56
Annelida	*Chone mollis*	–	–	1	–	–	1	0.16	98.72
Crustacea	Cumacea	–	–	1	–	–	1	0.16	98.88
Bivalvia	*Pandora bilirata*	–	–	–	1	–	1	0.16	99.04
Annelida	*Nephtys cornuta*	–	–	–	1	–	1	0.16	99.20
Annelida	*Tenonia priops*	–	–	–	1	–	1	0.16	99.36
Crustacea	*Cancer*	–	–	–	1	–	1	0.16	99.52
Mollusca	Gastropoda	–	–	–	1	–	1	0.16	99.68
Mollusca	*Polinices lewisii*	–	–	–	–	1	1	0.16	99.84
Annelida	*Axiothella rubrocincta*	–	–	–	–	1	1	0.16	100.00
Total Number of Individuals		**94**	**164**	**131**	**138**	**96**	**623**		

Table E.10 Species Abundance within Replicate Samples Collected at Benthic Station 9 in October 2005

Taxonomic Group	Taxon	Replicate 1	2	3	4	5	Total	Percent of Total	Cumulative Percent
Crustacea	*Photis macinerneyi*	27	23	15	36	47	148	24.79	24.79
Crustacea	*Eohaustorius sencillus*	16	16	3	23	17	75	12.56	37.35
Crustacea	*Rhepoxynius menziesi*	10	10	9	12	12	53	8.88	46.23
Nemertea	*Carinoma mutabilis*	9	5	14	7	9	44	7.37	53.60
Crustacea	*Majoxiphalus major*	5	11	9	10	2	37	6.20	59.80
Crustacea	*Rhepoxynius abronius*	4	5	3	6	4	22	3.69	63.48
Bivalvia	*Tellina bodegensis*	–	3	3	9	6	21	3.52	67.00
Annelida	*Scoloplos armiger*	2	1	5	6	5	19	3.18	70.18
Echinodermata	*Dendraster excentricus*	1	5	1	2	10	19	3.18	73.37
Crustacea	*Tecticeps convexus*	1	9	–	3	5	18	3.02	76.38
Crustacea	*Pacifoculodes spinipes*	5	1	2	5	3	16	2.68	79.06
Annelida	*Nephtys caecoides*	1	12	1	1	1	16	2.68	81.74
Bivalvia	*Tellina modesta*	2	–	2	4	4	12	2.01	83.75
Echinodermata	*Amphiodia*	5	–	–	3	3	11	1.84	85.59
Annelida	*Glycinde armigera*	2	–	–	3	4	9	1.51	87.10
Annelida	*Glycera convoluta*	1	–	6	–	2	9	1.51	88.61
Crustacea	*Americhilidium shoemakeri*	1	3	–	2	2	8	1.34	89.95
Annelida	*Chaetozone setosa*	3	2	1	–	1	7	1.17	91.12
Bivalvia	*Siliqua lucida*	1	3	1	1	1	7	1.17	92.29
Bivalvia	*Nuculana*	–	–	1	3	2	6	1.01	93.30
Crustacea	*Mesolamprops dillonensis*	–	1	2	1	–	4	0.67	93.97
Annelida	*Ampharete labrops*	–	1	2	–	1	4	0.67	94.64
Bivalvia	*Cooperella subdiaphana*	–	1	–	2	1	4	0.67	95.31
Annelida	*Apoprionospio pygmaea*	1	–	–	1	1	3	0.50	95.81
Mollusca	*Turbonilla*	–	1	1	–	–	2	0.34	96.15
Nemertea	*Cerebratulus californiensis*	–	1	1	–	–	2	0.34	96.48
Annelida	*Lumbrineris californiensis*	–	–	2	–	–	2	0.34	96.82
Annelida	*Onuphis iridescens*	–	–	2	–	–	2	0.34	97.15
Crustacea	*Anchicolurus occidentalis*	–	–	1	1	–	2	0.34	97.49
Nemertea	Hoplonemertea	–	–	–	–	2	2	0.34	97.82
Bivalvia	*Rochefortia tumida*	–	–	–	–	2	2	0.34	98.16
Crustacea	*Photis brevipes*	1	–	–	–	–	1	0.17	98.32
Annelida	*Diopatra ornata*	1	–	–	–	–	1	0.17	98.49
Crustacea	*Synidotea bicuspida*	1	–	–	–	–	1	0.17	98.66
Annelida	*Sigalion spinosa*	–	1	–	–	–	1	0.17	98.83
Mollusca	*Polinices lewisii*	–	1	–	–	–	1	0.17	98.99
Annelida	*Magelona sacculata*	–	1	–	–	–	1	0.17	99.16
Crustacea	*Gibberosus myersi*	–	–	1	–	–	1	0.17	99.33
Nemertea	*Micrura alaskensis*	–	–	1	–	–	1	0.17	99.50
Mollusca	*Rictaxis punctocaelatus*	–	–	1	–	–	1	0.17	99.66

Table E.10 Species Abundance within Replicate Samples Collected at Benthic Station 9 in October 2005

Taxonomic Group	Taxon	Replicate 1	2	3	4	5	Total	Percent of Total	Cumulative Percent
Crustacea	*Edotia sublittoralis*	–	–	–	–	1	1	0.17	99.83
Annelida	*Chone mollis*	–	–	–	–	1	1	0.17	100.00
Total Number of Individuals		**100**	**117**	**90**	**141**	**149**	**597**		

Table E.11 Infaunal Indices in Sediment Samples Collected on 4 October 2005

Station	Parameter[a]	Total Organisms	Identified Organisms	Number of Species	Organisms per Species	Diversity (H')	Brillouin Index	Dominance (C')	Dominance (75)	Species Richness (d)	Evenness (J')	Infaunal Index
Station 1	Replicate 1	117	98	26	3.77	2.86	2.52	0.08	11	5.45	0.88	90
	Replicate 2	157	150	26	5.77	2.75	2.50	0.09	9	4.99	0.84	81
	Replicate 3	122	119	32	3.72	2.86	2.52	0.10	11	6.49	0.82	87
	Replicate 4	146	138	35	3.94	2.97	2.65	0.08	12	6.90	0.84	76
	Replicate 5	75	73	20	3.65	2.44	2.11	0.13	7	4.43	0.82	91
Station 2	Replicate 1	109	106	26	4.08	2.75	2.43	0.10	10	5.36	0.85	79
	Replicate 2	81	78	21	3.71	2.44	2.11	0.14	7	4.59	0.80	89
	Replicate 3	93	92	22	4.18	2.66	2.35	0.09	8	4.64	0.86	78
	Replicate 4	57	57	16	3.56	2.52	2.17	0.10	8	3.71	0.91	88
	Replicate 5	42	40	17	2.35	2.55	2.08	0.10	8	4.34	0.90	80
Station 3	Replicate 1	69	67	18	3.72	2.61	2.26	0.09	9	4.04	0.90	86
	Replicate 2	88	88	21	4.19	2.62	2.31	0.10	9	4.47	0.86	76
	Replicate 3	61	60	18	3.33	2.39	2.04	0.13	7	4.15	0.83	84
	Replicate 4	73	69	17	4.06	2.49	2.17	0.11	8	3.78	0.88	83
	Replicate 5	89	88	19	4.63	2.56	2.27	0.10	8	4.02	0.87	83
Station 4	Replicate 1	86	82	27	3.04	2.86	2.46	0.08	11	5.90	0.87	83
	Replicate 2	79	78	21	3.71	2.62	2.28	0.11	9	4.59	0.86	82
	Replicate 3	77	76	23	3.30	2.74	2.36	0.09	9	5.08	0.87	87
	Replicate 4	109	105	31	3.39	3.10	2.71	0.06	14	6.45	0.90	83
	Replicate 5	90	88	22	4.00	2.29	1.99	0.20	7	4.69	0.74	82
Station 5	Replicate 1	117	116	23	5.04	2.62	2.35	0.10	7	4.63	0.83	82
	Replicate 2	94	90	27	3.33	3.01	2.62	0.06	12	5.78	0.91	79
	Replicate 3	123	119	25	4.76	2.68	2.40	0.10	9	5.02	0.83	78
	Replicate 4	64	60	27	2.22	2.94	2.44	0.08	12	6.35	0.89	82
	Replicate 5	88	86	26	3.31	2.99	2.60	0.06	12	5.61	0.92	80
Station 6	Replicate 1	110	108	25	4.32	2.68	2.39	0.10	8	5.13	0.83	85
	Replicate 2	176	165	29	5.69	2.70	2.46	0.10	8	5.48	0.80	74
	Replicate 3	138	131	24	5.46	2.58	2.33	0.11	8	4.72	0.81	72
	Replicate 4	156	143	27	5.30	2.57	2.32	0.13	8	5.24	0.78	76
	Replicate 5	121	113	25	4.52	2.68	2.38	0.11	9	5.08	0.83	81
Station 7	Replicate 1	100	97	21	4.62	2.55	2.26	0.12	8	4.37	0.84	90
	Replicate 2	113	108	21	5.14	2.39	2.13	0.14	6	4.27	0.78	75
	Replicate 3	98	92	27	3.41	2.66	2.31	0.11	9	5.75	0.81	74
	Replicate 4	97	93	23	4.04	2.76	2.43	0.08	9	4.85	0.88	82
	Replicate 5	98	96	22	4.36	2.76	2.45	0.08	10	4.60	0.89	80
Station 8	Replicate 1	94	84	24	3.50	2.85	2.49	0.07	11	5.19	0.90	79
	Replicate 2	164	157	35	4.49	2.99	2.69	0.08	12	6.72	0.84	80
	Replicate 3	131	127	30	4.23	2.78	2.48	0.09	9	5.99	0.82	87
	Replicate 4	138	129	33	3.91	2.92	2.59	0.08	11	6.58	0.84	79
	Replicate 5	96	94	25	3.76	2.63	2.31	0.11	8	5.28	0.82	85
Station 9	Replicate 1	100	95	21	4.52	2.39	2.12	0.14	6	4.39	0.79	80
	Replicate 2	117	116	22	5.27	2.59	2.33	0.10	8	4.42	0.84	80
	Replicate 3	90	88	24	3.67	2.73	2.38	0.09	9	5.14	0.86	81
	Replicate 4	141	135	20	6.75	2.44	2.22	0.13	7	3.87	0.81	78
	Replicate 5	149	142	24	5.92	2.46	2.23	0.15	8	4.64	0.78	75

Table E.11 Infaunal Indices in Sediment Samples Collected on 4 October 2005

	Parameter[a]	Total Organisms	Identified Organisms	Number of Species	Organisms per Species	Diversity (H')	Brillouin Index	Dominance (C')	Dominance (75)	Species Richness (d)	Evenness (J')	Infaunal Index
Mean	Station 1	123	116	27.8	4.2	2.78	2.46	0.10	10.0	5.65	0.84	85
	Station 2	76	75	20.4	3.6	2.58	2.23	0.11	8.2	4.53	0.86	83
	Station 3	76	74	18.6	4.0	2.53	2.21	0.11	8.2	4.09	0.87	82
	Station 4	88	86	24.8	3.5	2.72	2.36	0.11	10.0	5.34	0.85	83
	Station 5	97	94	25.6	3.7	2.85	2.48	0.08	10.4	5.48	0.88	80
	Station 6	140	132	26.0	5.1	2.64	2.38	0.11	8.2	5.13	0.81	78
	Station 7	101	97	22.8	4.3	2.62	2.32	0.11	8.4	4.77	0.84	80
	Station 8	125	118	29.4	4.0	2.84	2.51	0.09	10.2	5.95	0.84	82
	Station 9	119	115	22.2	5.2	2.52	2.26	0.12	7.6	4.49	0.81	79
Standard Deviation	Station 1	32	31	5.8	0.90	0.20	0.20	0.02	2.0	1.03	0.02	6.6
	Station 2	27	26	4.0	0.73	0.12	0.16	0.02	1.1	0.59	0.04	5.3
	Station 3	12	13	1.5	0.49	0.09	0.11	0.02	0.8	0.25	0.03	3.9
	Station 4	13	12	4.1	0.37	0.30	0.27	0.05	2.6	0.81	0.06	1.8
	Station 5	24	24	1.7	1.16	0.18	0.12	0.02	2.3	0.67	0.04	1.7
	Station 6	27	23	2.0	0.60	0.06	0.05	0.01	0.4	0.28	0.02	5.4
	Station 7	7	6	2.5	0.65	0.16	0.13	0.03	1.5	0.59	0.05	6.4
	Station 8	30	29	4.8	0.39	0.14	0.14	0.01	1.6	0.71	0.03	3.7
	Station 9	25	24	1.8	1.20	0.14	0.10	0.02	1.1	0.46	0.03	2.4
Station Total	Station 1	617	578	48	12.04	3.06	2.92	0.08	12	7.39	0.79	84
	Station 2	382	373	39	9.56	2.90	2.74	0.08	10	6.42	0.79	82
	Station 3	380	372	28	13.29	2.84	2.71	0.07	10	4.56	0.85	82
	Station 4	441	429	43	9.98	3.02	2.86	0.08	12	6.93	0.80	83
	Station 5	486	471	43	10.95	3.08	2.93	0.07	12	6.82	0.82	80
	Station 6	701	660	48	13.75	3.00	2.88	0.08	11	7.24	0.78	78
	Station 7	506	486	42	11.57	2.85	2.72	0.09	9	6.63	0.76	80
	Station 8	623	591	52	11.37	3.12	2.97	0.07	12	7.99	0.79	81
	Station 9	597	576	38	15.16	2.71	2.60	0.11	9	5.82	0.75	79
Survey	**Total**	**4733**	**4536**	**83**	**54.65**	**3.12**	**3.08**	**0.07**	**12**	**9.74**	**0.71**	**81**

[a] Except for Total Organisms and the Infaunal Index, all parameters are computed using only those organisms identified to species level. One specimen of *Pisaster brevispinus* collected in Replicate Sample 5 at Station 2 was excluded from the infaunal analysis because it is undersampled epifauna.

Table E.12 Infaunal Indices without *D. excentricus* in Sediment Samples Collected on 4 October 2005

	Parameter	Total Organisms	Identified Organisms	Number of Species	Organisms per Species	Diversity (H')	Brillouin Index	Dominance (C')	Dominance (75)	Species Richness (d)	Evenness (J')	Infaunal Index
Station 1	Replicate 1	114	95	25	3.80	2.81	2.47	0.08	10	5.27	0.87	90
	Replicate 2	149	142	25	5.68	2.69	2.44	0.10	9	4.84	0.84	81
	Replicate 3	115	112	31	3.61	2.80	2.45	0.11	10	6.36	0.81	87
	Replicate 4	142	134	34	3.94	2.93	2.60	0.08	12	6.74	0.83	76
	Replicate 5	72	70	19	3.68	2.37	2.04	0.14	7	4.24	0.80	91
Station 2	Replicate 1	106	103	25	4.12	2.70	2.39	0.11	9	5.18	0.84	79
	Replicate 2	79	76	20	3.80	2.38	2.06	0.15	7	4.39	0.79	89
	Replicate 3	86	85	21	4.05	2.59	2.27	0.10	8	4.50	0.85	78
	Replicate 4	55	55	15	3.67	2.46	2.11	0.10	8	3.49	0.91	88
	Replicate 5	39	37	16	2.31	2.47	2.00	0.11	8	4.15	0.89	80
Station 3	Replicate 1	69	67	18	3.72	2.61	2.26	0.09	9	4.04	0.90	86
	Replicate 2	81	81	20	4.05	2.55	2.23	0.11	8	4.32	0.85	76
	Replicate 3	59	58	17	3.41	2.32	1.98	0.14	6	3.94	0.82	84
	Replicate 4	70	66	16	4.13	2.42	2.10	0.11	7	3.58	0.87	83
	Replicate 5	87	86	18	4.78	2.51	2.23	0.11	8	3.82	0.87	83
Station 4	Replicate 1	81	77	26	2.96	2.80	2.40	0.09	10	5.76	0.86	83
	Replicate 2	79	78	21	3.71	2.62	2.28	0.11	9	4.59	0.86	82
	Replicate 3	77	76	23	3.30	2.74	2.36	0.09	9	5.08	0.87	87
	Replicate 4	107	103	30	3.43	3.06	2.68	0.06	13	6.26	0.90	83
	Replicate 5	87	85	21	4.05	2.22	1.92	0.21	6	4.50	0.73	82
Station 5	Replicate 1	115	114	22	5.18	2.57	2.31	0.10	7	4.43	0.83	82
	Replicate 2	90	86	26	3.31	2.96	2.57	0.06	11	5.61	0.91	79
	Replicate 3	121	117	24	4.88	2.64	2.37	0.10	8	4.83	0.83	78
	Replicate 4	62	58	26	2.23	2.89	2.39	0.08	12	6.16	0.89	82
	Replicate 5	84	82	25	3.28	2.94	2.54	0.07	11	5.45	0.91	80
Station 6	Replicate 1	108	106	24	4.42	2.64	2.35	0.10	8	4.93	0.83	85
	Replicate 2	164	153	28	5.46	2.63	2.39	0.12	8	5.37	0.79	74
	Replicate 3	134	127	23	5.52	2.52	2.28	0.12	7	4.54	0.81	72
	Replicate 4	154	141	26	5.42	2.53	2.29	0.13	8	5.05	0.78	76
	Replicate 5	118	110	24	4.58	2.63	2.34	0.11	8	4.89	0.83	81
Station 7	Replicate 1	97	94	20	4.70	2.49	2.21	0.12	8	4.18	0.83	90
	Replicate 2	111	106	20	5.30	2.34	2.09	0.14	6	4.07	0.78	75
	Replicate 3	95	89	26	3.42	2.61	2.26	0.12	8	5.57	0.80	74
	Replicate 4	95	91	22	4.14	2.72	2.40	0.09	9	4.66	0.88	82
	Replicate 5	93	91	21	4.33	2.70	2.39	0.09	9	4.43	0.89	80
Station 8	Replicate 1	91	81	23	3.52	2.80	2.44	0.08	10	5.01	0.89	79
	Replicate 2	162	155	34	4.56	2.96	2.66	0.08	11	6.54	0.84	80
	Replicate 3	117	113	29	3.90	2.74	2.42	0.10	9	5.92	0.81	87
	Replicate 4	130	121	32	3.78	2.87	2.53	0.09	10	6.46	0.83	79
	Replicate 5	92	90	24	3.75	2.57	2.24	0.12	7	5.11	0.81	85
Station 9	Replicate 1	99	94	20	4.70	2.36	2.09	0.14	6	4.18	0.79	80
	Replicate 2	112	111	21	5.29	2.52	2.27	0.11	7	4.25	0.83	80
	Replicate 3	89	87	23	3.78	2.69	2.36	0.09	9	4.93	0.86	81
	Replicate 4	139	133	19	7.00	2.40	2.19	0.13	7	3.68	0.81	78
	Replicate 5	139	132	23	5.74	2.38	2.14	0.17	7	4.51	0.76	75

Table E.12 Infaunal Indices without *D. excentricus* in Sediment Samples Collected on 4 October 2005

	Parameter	Total Organisms	Identified Organisms	Number of Species	Organisms per Species	Diversity (H')	Brillouin Index	Dominance (C')	Dominance (75)	Species Richness (d)	Evenness (J')	Infaunal Index
Mean	Station 1	118	111	26.8	4.1	2.72	2.40	0.10	9.6	5.49	0.83	85
	Station 2	73	71	19.4	3.6	2.52	2.17	0.11	8.0	4.34	0.86	83
	Station 3	73	72	17.8	4.0	2.48	2.16	0.11	7.6	3.94	0.86	82
	Station 4	86	84	24.2	3.5	2.69	2.33	0.11	9.4	5.24	0.84	83
	Station 5	94	91	24.6	3.8	2.80	2.44	0.08	9.8	5.30	0.87	80
	Station 6	136	127	25.0	5.1	2.59	2.33	0.11	7.8	4.96	0.81	78
	Station 7	98	94	21.8	4.4	2.57	2.27	0.11	8.0	4.58	0.84	80
	Station 8	118	112	28.4	3.9	2.79	2.46	0.09	9.4	5.81	0.84	82
	Station 9	116	111	21.2	5.3	2.47	2.21	0.13	7.2	4.31	0.81	79
Standard Deviation	Station 1	30	29	5.8	0.87	0.21	0.21	0.02	1.8	1.04	0.03	6.6
	Station 2	26	26	4.0	0.74	0.13	0.16	0.02	0.7	0.61	0.04	5.3
	Station 3	11	12	1.5	0.51	0.11	0.12	0.02	1.1	0.27	0.03	3.9
	Station 4	12	11	3.8	0.41	0.31	0.27	0.06	2.5	0.76	0.07	1.8
	Station 5	24	24	1.7	1.23	0.18	0.11	0.02	2.2	0.68	0.04	1.7
	Station 6	24	20	2.0	0.53	0.06	0.04	0.01	0.4	0.30	0.02	5.4
	Station 7	7	7	2.5	0.69	0.16	0.13	0.03	1.2	0.60	0.05	6.4
	Station 8	30	29	4.8	0.39	0.15	0.16	0.02	1.5	0.72	0.03	3.7
	Station 9	23	21	1.8	1.20	0.14	0.11	0.03	1.1	0.46	0.04	2.4
Station Total	Station 1	592	553	47	11.77	3.02	2.87	0.08	12	7.28	0.78	84
	Station 2	365	356	38	9.37	2.84	2.68	0.08	9	6.30	0.78	82
	Station 3	366	358	27	13.26	2.78	2.65	0.08	9	4.42	0.85	82
	Station 4	431	419	42	9.98	2.98	2.82	0.09	11	6.79	0.80	83
	Station 5	472	457	42	10.88	3.04	2.89	0.07	11	6.69	0.81	80
	Station 6	678	637	47	13.55	2.96	2.83	0.08	10	7.12	0.77	78
	Station 7	491	471	41	11.49	2.80	2.67	0.09	8	6.50	0.76	80
	Station 8	592	560	51	10.98	3.08	2.93	0.08	12	7.90	0.78	81
	Station 9	578	557	37	15.05	2.66	2.55	0.12	8	5.69	0.74	79
Survey	**Total**	**4565**	**4368**	**82**	**53.27**	**3.08**	**3.04**	**0.07**	**11**	**9.66**	**0.70**	**81**